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MOUNT POLLEY REHABILITATION AND REMEDIATION STRATEGY

Ecological Risk Assessment

ECOLOGICAL RISK ASSESSMENT

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Executive Summary

The Mount Polley Mine is an open-pit and underground copper and gold mine near Likely, British Columbia. On 4 August 2014, the failure of a glacial lacustrine layer beneath the Perimeter Embankment of the Tailings Storage Facility (“the breach”) at the mine resulted in the release of a slurry of water, tailings, and dam construction material. The material resulted in physical, chemical, and biological impacts to Polley Lake, Hazeltine and Edney Creeks and Quesnel Lake. The impacts of the breach were evaluated in a series of technical studies and data summaries in a Post Event Environmental Impact Assessment Report, which was updated as additional data became available. Further studies were also subsequently undertaken to monitor the reconstruction of the Hazeltine Channel and to better understand potential environmental risks associated with the deposited material.

Risk Assessment Approach

An ecological risk assessment determines the ecological significance of environmental changes. Changes to the environment include metals associated with tailings and physical impacts associated with the breach. Risk assessment is part of the overall remediation strategy that Mount Polley Mining Corporation developed in response to the breach. The long-term goal is to re-establish a biologically diverse, functional and self-sustaining ecosystem in the areas impacted by the breach. The focus of the risk assessment was to integrate the available information to determine if additional rehabilitation work is necessary to address residual contamination. The implications for longer-term recovery was evaluated as appropriate.

Risk assessment is inherently cautionary and its application to this site incorporated numerous conservative approaches, assumptions, and calculations to avoid inadvertently concluding that risks are not present if they actually are. The risk assessment considered the concentrations of metals measured in soil, sediment and water samples from the site. Copper, arsenic, or vanadium (depending on the media) were identified as the primary contaminants of potential concern because concentrations were greater than the applicable regulatory guideline values. However, concentrations greater than a guideline value does not necessarily mean that environmental consequences (risk) is present. For instance, geochemical studies of sediment samples collected as part of the impact assessment indicated that elevated arsenic values were not associated with the tailings, but reflect a natural elevation in arsenic in the area. Other types of information that were considered in addition to the chemistry data (which provides information about exposure) included experimental data (e.g., toxicity testing that measured potential effects in a laboratory setting) and field data (e.g., in-situ measurements and observations that measure potential effects at the actual site). A weight-of-evidence approach was used to integrate these three main types of information into an overall conclusion.

Terrestrial Risk Assessment

The terrestrial risk assessment focused on the Hazeltine Creek corridor which consisted of the “floodplain” area where the forest and portions of native soils were removed by the debris flow and the “halo” area where the forest floor was relatively undisturbed, but was buried to varying depths by scoured material and tailings. The contaminants of potential concern in soil were copper and vanadium. The risk assessment evaluated risks to soil microbes and invertebrates, plants, and wildlife by considering soil concentrations, soil geochemistry, invertebrate and plant tissue concentrations, plant toxicity testing, forest health observations such as the type and abundance of soil fungi and changes in tree health, and food-chain modelling to estimate hazards to terrestrial birds and mammals.

Although concentrations of copper and vanadium in soil exceeded numerical standards for the protection of soil invertebrates and plants, there were multiple lines of evidence in the geochemistry work that demonstrated that the tailings are not acid generating and have a low potential for leaching of metals. Concentrations of metals in soil porewater or groundwater are expected to remain stable or decrease over time. Metals are expected to remain part of the minerals in the tailings and would therefore be likely to have low bioavailability to plants, invertebrates or wildlife. The plant toxicity testing and tissue chemistry (plants and soil invertebrates) confirmed the geochemical assessment that copper and vanadium would be expected to have low bioavailability. The plant toxicity testing also helped to identify the most likely cause of the impacts observed in the forest health assessment. For example, many of the trees in the halo area that initially survived the breach died during the spring of 2015, which was attributed to a physical root smothering effect. Root decay and a lack of soil organisms (like fungi) that make up a “healthy” soil community suggested that the deposit of wet tailings had reduced soil aeration which led to the eventual mortality of the trees affected by those tailings. Food chain modeling was conducted to determine if the cumulative dose of copper and vanadium ingested by different wildlife species (i.e., the sum of food, water and soil) exceeded a benchmark value. The cumulative dose was lower than a conservative benchmark from the scientific literature for most of the wildlife species evaluated, and further studies to improve the understanding of hazards for the remaining species (primarily, birds and mammals that would consume large amounts of soil invertebrates) will be conducted.

Overall, risks associated with copper and vanadium in soils are considered to be low. A central observation that was identified in multiple lines of evidence was that the total concentration of metals in soil is not indicative of the environmental risk because the bioavailability of metals from tailings was shown to be low. Physical effects associated with the tailings (e.g., low nutrients, low moisture content) and habitat alteration are likely to be a more dominant stressor than the residual concentrations of metals. Rehabilitation to improve these physical and structural soil deficiencies is underway, and both natural and facilitated re-vegetation is occurring.

Aquatic Risk Assessment

The aquatic risk assessment focused on Polley Lake, Hazeltine Creek, Quesnel Lake, and Quesnel River. The contaminants of potential concern in sediment were copper and arsenic, and the contaminant of potential concern in water was copper. Arsenic was conservatively included as a contaminant of potential concern despite the geochemical assessment that concluded that it may be naturally elevated in sediment. The risk assessment evaluated risks to aquatic plants, aquatic invertebrates (both sediment-dwelling and those that live in the water column) and fish by considering water and sediment chemistry; toxicity testing; invertebrate and fish tissue chemistry; invertebrate and fish community structure; fish health metrics and food-chain modelling to estimate hazards to birds and mammals that consume fish.

Copper concentrations in Hazeltine Creek and localized areas in Quesnel Lake was higher than the water quality guideline for the protection of aquatic life following the breach in 2014. These elevated copper concentrations were in large part due to association with elevated concentrations of suspended solids. Copper concentrations gradually decreased through 2015 to below the guideline in all water bodies with the exception of Hazeltine Channel. Concentrations of copper in water samples from Hazeltine Channel exceeded water quality guidelines in 2015 and 2016. During this time, remediation and reconstruction activities were underway for much of the Hazeltine Channel which limited the opportunity to collect other lines of evidence. Water quality monitoring continued in 2017 and the results will be described in the monitoring program. The remediation and reconstruction work in upper Hazeltine Channel has been completed and biological monitoring is planned to provide context to the observed water concentrations to determine if further action is warranted.

In Polley and Quesnel Lakes, water concentrations of copper were at or less than the water quality guideline, and therefore, long-term effects to plants, water-column invertebrates and fish are not expected. This conclusion was confirmed by toxicity testing and field measurements (e.g., measurements of the abundance and diversity of plankton; fish size). Risks to fish from consuming prey items is considered to be low, as are risks to fish-eating mammals and birds.

The benthic environment was primarily affected by the deposit of tailings and debris from the scouring of the Hazeltine Corridor. There are impacts to the benthic community from this physical event. The risk assessment for the benthic communities focused on determining if copper concentrations in sediment presented additional risks that would prevent reestablishment of a biologically diverse, functional and self-sustaining ecosystem. Toxicity testing and benthic community assessment (including a transplant study) was completed. Overall, it is possible that copper may affect the benthic community (in particular, the deeper portion of Quesnel Lake); however, the available information indicates that this potential effect is relatively small compared to physical factors such as low organic carbon (i.e., nutrient deficiency). Copper is not expected to be a limiting factor for the natural recovery of the benthic community.

Conclusion

The results of this ecological assessment indicate that risks associated with metals in material released by the breach are acceptable, and of low magnitude relative to the physical effects. These findings are based on several studies conducted over multiple years since the breach occurred. Conditions in both the terrestrial and aquatic environments are considered to be stable or improving. As with any risk assessment process, there were areas of uncertainty identified, but all areas of uncertainty identified can be addressed as part of the ongoing environmental monitoring and recovery program by Mount Polley Mining Corporation.

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APPENDICES (SEPARATE)

List of Acronyms

AVS	Acid volatile sulphides
AVS-SEM	acid-volatile sulfide-simultaneously extracted metals
BC	British Columbia
BC MFR	British Columbia Ministry of Forests and Range
BC MoE	British Columbia Ministry of Environment and Climate Change Strategy
CCME	Canadian Council of Ministers of the Environment
COPC	contaminant of potential concern
CRC-ICP-MS	Collision cell inductively coupled plasma-mass spectrometry
CSR	Contaminated Sites Regulation
CV	Coefficient of variation
Cu	Copper
CVAFS	cold vapour atomic fluorescence spectrophotometry
DFO	Fisheries and Oceans Canada
DGT	Diffusive gradient—thin film device
DQO	Data quality objective
DSI	Detailed Site Investigation
DTPA	diethylenetriaminepentaacetiac acid
EC_X	Adverse effect concentration – x% adverse effect size; contrast with IC_x
e.g.	for example
EEM	Environmental Effects Monitoring
Golder	Golder Associates Ltd.
HAC	Hazeltine Creek
HR-ICP-MS	high resolution inductively coupled plasma mass spectrometry
i.e.	that is
IC25	The 25th percentile inhibitory concentration
ICP-MS	inductively coupled plasma mass spectrometry
LD_{50}	Tissue concentration that results in 50% effect
LC_{50}	Lethal concentration – 50% adverse effects
LOE	line of evidence
MFR	Ministry of Forest and Resources
MPMC	Mount Polley Mining Corporation
PBET	Physiologically based extraction test
PEEIAR	Post Event Environmental Impact Assessment Report
PEL	Probable Effect Level (CCME sediment quality guideline)
PHREEQC	pH Redox Equilibrium in C language

QA/QC	Quality assurance/quality control
QUL	Quesnel Lake
QUR	Quesnel River
RPD	Relative percent difference
SABCS	Science Advisory Board for Contaminated Sites
SARA	*Species at Risk Act*
$SedQC_{SS}$	Sediment Quality Criterion (sensitive contaminated site)
$SedQC_{TS}$	Sediment Quality Criterion (typical contaminated site)
SEM	Simultaneously extractable metals
sp.	unspecified species
spp.	several species
SRK	SRK Consulting
TOC	total organic carbon
TRV	toxicity reference value
TSF	tailings storage facility
TSS	total suspended solids
UBC	University of British Columbia
UCLM	upper confidence limits of the mean
US EPA	United States Environmental Protection Agency
WOE	weight-of-evidence
WQG	water quality guideline

1.0 INTRODUCTION

1.1 Purpose of the Risk Assessment

Golder Associates Ltd. (Golder) was retained by Mount Polley Mining Corporation (MPMC) to conduct an ecological risk assessment as required by Pollution Abatement Order #107461 issued on 9 June 2017 (amending and consolidating previous Orders) pursuant to Section 83 (1c) of the *Environmental Management Act* (EMA). The human health risk assessment required by the Order has been submitted to the Ministry of Environment and Climate Change Strategy (MoE) as a separate document and was accepted (as required by the Order).

The purpose of this ecological risk assessment (ERA) is to determine the ecological significance of altered environmental conditions following the Tailing Storage Facility embankment breach (the 'TSF embankment breach'). Essentially, this requires determining if the residual concentrations of metals in tailings presents an unacceptable risk that requires risk management over and above the rehabilitation and monitored recovery program that is already underway to address the physical stressors (nutrient and physical deficiencies) that were inherent from the scouring and deposition event. An unacceptable risk, in this context, is related to whether there are any effects from chemical impacts related to the TSF embankment breach that will prevent the long-term successful re-introduction of a biologically diverse, functional, self-sustaining and inter-dependent ecosystem.

The results of this risk assessment will assist MPMC in determining an appropriate course of action for long-term management of Site conditions by informing the development of the Remediation Plan.

1.1 Project Context

A detailed description of the physical, chemical, and biological alterations observed following the TSF embankment breach was provided in the Post-Event Environmental Impact Assessment Report (PEEIAR; MPMC 2015). In brief, the TSF embankment breach caused water and tailings to be released to Polley Lake, the former Hazeltine Creek, and Quesnel Lake. This release eroded the slopes adjacent to the former Hazeltine Creek (now called Hazeltine Channel; i.e., the totality of upper Hazeltine Creek, Hazeltine Canyon and lower Hazeltine Creek shown on Figure 2), removing the forest vegetation and native soil; the channel bed and associated substrate along the creek channel is now referred to as the "floodplain". The channel bed was eroded to bedrock in some locations. Approximately 136 hectares (ha) (1.36 square kilometres) was scoured of forest and topsoil, and tailings were deposited above relatively undisturbed forest floor across an additional 100 ha (1 square kilometres). A mixture of tailings and scoured native material was transported to Polley and Quesnel Lakes. Approximately 12.8 million cubic metres (M m^3) of tailings (plus an additional 5.8 M m^3 of native soil and TSF water) was deposited to Quesnel Lake (MPMC 2015).

The environmental rehabilitation strategy for the TSF breach consists of four main phases (Figure 1) over nine identified remediation areas (Figure 2) that were developed early in the response and that have been followed since. Conceptually, these phases are presented sequentially with the information generated or decisions made in one phase feeding into the planning, actions or definition of information needs for a subsequent phase. However, given the dynamic nature of the response and conditions under which it was implemented, some of the actions required resequencing, but are nevertheless consistent with the framework (Nikl et al. 2016). The need for an adaptive response was also a foreseeable necessity for an incident of this nature—the regulatory instruments (e.g., the Order, instructions issued pursuant to the Order) were themselves adaptive, and amended orders were issued by government during the response as physical works progressed and more data became available.

Risk management/rehabilitation activities have been underway since the TSF embankment breach and the physical impacts associated with the initial release are being addressed through a broad and consultative habitat rehabilitation program that is directed towards a return to productivity of those habitats. Initiation of these works (which would be part of Phase 4) has preceded the conclusion of this risk assessment (Phase 3) because doing so was a necessary part of responding to the Order's requirement to cease discharge of turbid waters and because Government, First Nations, Stakeholders and the Mine desired return of habitats to productive use as early as possible.

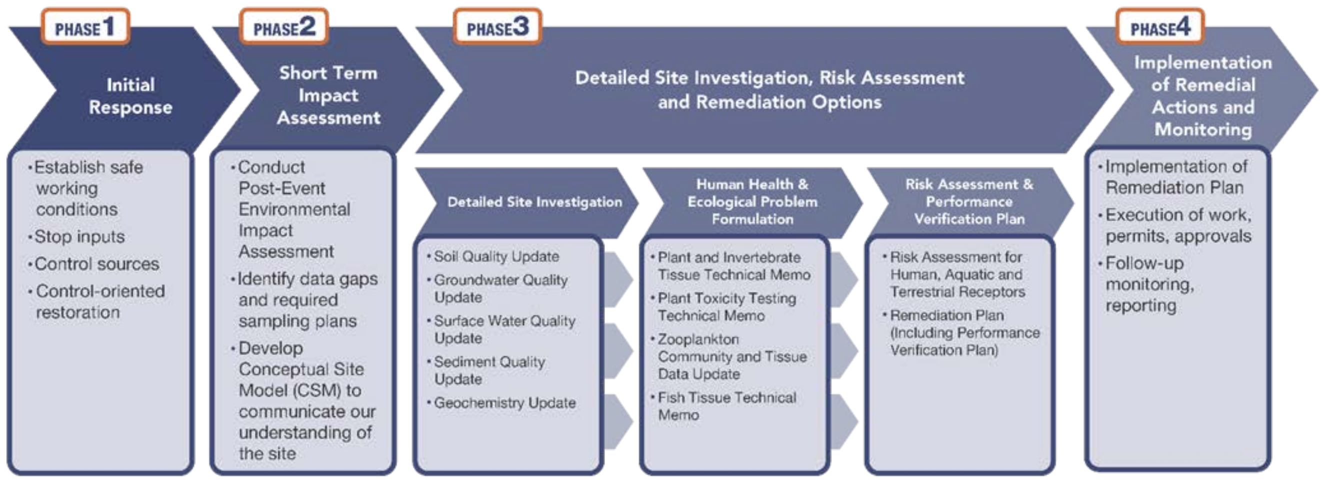

Figure 1: The Adaptive Rehabilitation Framework Used in the Response to the Embankment Breach

As of the time of writing, the status of progress on the rehabilitation strategy is as follows:

- Phase I (Initial Response) has been concluded with the Hazeltine Channel, channel reconstruction and armouring now complete (Bronsro et al., 2016) and the TSF embankment repaired and upgraded.
- Phase II (Short-Term Impact Assessment) has been concluded with the PEEIAR (MPMC 2015) and an update to the report (Golder 2016a) submitted to government and posted to the Imperial Metals web site.
- Phase III of the rehabilitation strategy, which is an active stage, included the preparation of a detailed site investigation (DSI; accepted by government), a human health risk assessment (accepted by government), and an ecological risk assessment (this document). This phase focuses on the long-term conditions that remain following the initial impact and implementation of short-term remedial measures (e.g., channel for the Hazeltine Channel/Edney Creek re-constructed and conveying water from the Site to Quesnel Lake). A draft annotated Table of Contents for a remediation plan has been submitted to government but the plan has not yet been prepared as it will be predicated to a considerable extent on the findings of both the Human and Ecological Risk Assessments.
- Phase IV of the rehabilitation strategy is focused on implementing additional remediation, as needed, following completion of the risk assessment. A comprehensive environmental monitoring plan, which is intended to harmonize breach, permit and *Metal Mining Effluent Regulations* mandated monitoring programs was developed by MPMC and is currently under review by government.

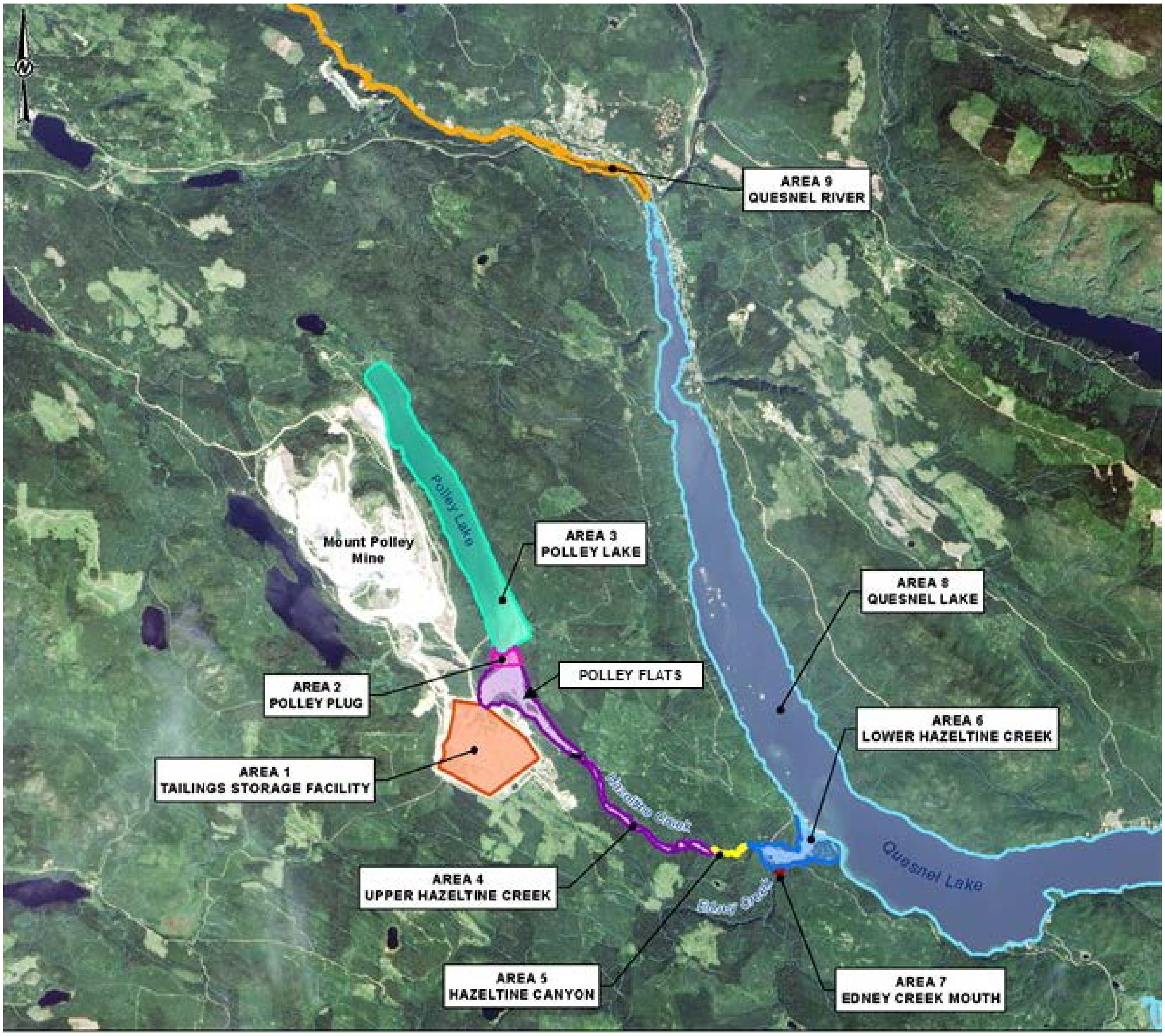

Figure 2: Overview of the Study Area and Remediation Areas

1.2 Technical Approach for Risk Assessment

1.2.1 Relevant Guidance Documents

The risk assessment was conducted in a manner compatible with relevant guidance and common practice in British Columbia. Key guidance documents are briefly summarized below:

- Provincial guidance - Procedures and policy decisions are primarily contained in Protocol 20 and Technical Guidance 7. Protocol 20 highlights the process of setting management objectives, assessment and measurement endpoints, identifying receptors of concern, and using weight-of-evidence approaches (Science Advisory Board for Contaminated Sites (SABCS) 2008; 2011)[1]. Technical Guidance 7 provides "guidance related to the performance of human health and ecological risk assessments for contaminated sites in British Columbia [to] supplement existing provisions in protocols under the Environmental Management Act". The guidance identifies preferred approaches and data sources for model parameterization, toxicity reference values, selection of toxicity tests and the use of weight-of-evidence.
- National and international guidance - There are numerous risk assessment guidance documents that have contributed to common practice (e.g., Environment Canada 2012a, United States Environmental Protection Agency (US EPA) 1998). These supplementary sources were used to guide decisions where not in conflict with the provincial protocols and procedures.

1.2.2 Approach to Risk Assessment

The common approach to risk assessment consists of problem formulation, exposure and effects assessment and risk characterization (SABCS 2008; Environment Canada 2012a). A brief overview of each of these major risk assessment components (Figure 3) is provided:

- Problem Formulation— The problem formulation entails initial screening of the three main components that need to co-occur to cause unacceptable risk (hazardous concentrations of chemicals, presence of sensitive receptors, and potentially significant exposure pathways). The problem formulation is a planning stage to identify the extent of the problem, establish the objectives of the risk assessment, and select appropriate approaches (SABCS 2008; Environment Canada 2012a). The problem formulation is summarized into a conceptual site model that highlights stressor sources, receptors of interest, and potentially significant exposure pathways; the conceptual site model considers the site investigation data and our understanding of the physicochemical properties and fate of the contaminants of potential concern (COPCs).
- Exposure and Effect Assessment — The exposure assessment provides an analysis of exposure condition or dose for each potentially significant exposure pathway identified in the problem formulation. For lower trophic level receptors, such as soil invertebrates and plants, exposure is often evaluated based on a concentration in soil, sediment, or water (mg/kg or mg/L). The term exposure point concentration is used to describe the degree of exposure to abiotic media that is aggregated to a scale relevant to the receptor, and that applies consideration of variability over space and time. For higher trophic level receptors, such as mammals and birds, exposure is usually estimated by a calculation of dose ($mg \cdot kg^{-1} \cdot d^{-1}$) received via ingestion of soil (or sediment), drinking water, and dietary items (plants and organism tissues). The effects assessment focused on determining the safe level of exposure or dose either through published literature sources or by direct measures of toxicity using bioassays.

[1] A completed Protocol 20 checklist has been included in an Appendix to this document.

- Risk Characterization— The risk characterization involves integrating the findings from the analysis phase to describe the risk in terms of its magnitude and type, and analyzing and discussing uncertainty in the process to provide an indication of the confidence in the risk assessment.

Risk assessment is often an iterative approach (generically described in stages such as screening, preliminary quantitative, detailed quantitative) where the decision to conduct another iteration (i.e., refinement of previous studies and conclusions) is based on the level of uncertainty in the risk estimate relative to the needs of the risk management planning process.

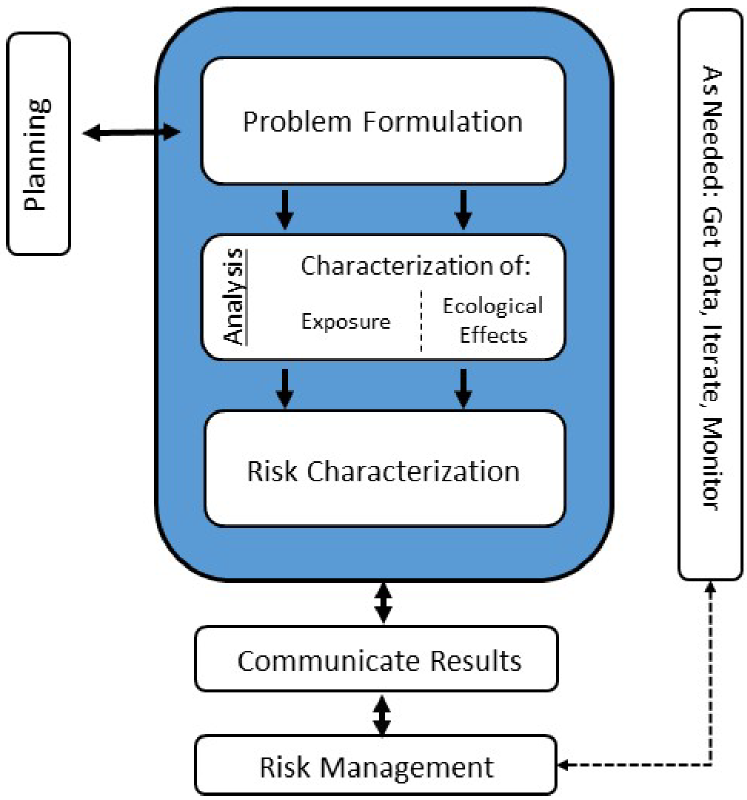

Figure 3: Overview of Major Phases of an Ecological Risk Assessment

(Adapted from: https://www.epa.gov/risk/conducting-ecological-risk-assessment).

Weight-of-evidence was the procedure used for the risk characterization step. The risk assessor uses a number of lines of evidence (i.e., measurement endpoints) that contribute to the overall understanding of risk related to contaminants or other stressors at the site. Different measurement endpoints convey different strength of information (e.g., statistical confidence, biological relevance, representation of causation). The measurement endpoints are linked to assessment endpoints that are the narrative statements on valued ecosystem components being protected (e.g., maintenance of healthy, diverse, and functional natural plant communities). The assessment endpoints are linked to the goals of the risk assessment, thus enabling the evaluation and success of the risk assessment in meeting study goals. Measurement and assessment endpoints and goals are further detailed in the problem formulation section of this report.

A customized weight-of-evidence approach was adapted for this assessment based on past experience, input from government, First Nations and stakeholders, and case studies from the scientific literature. The lines of evidence were based on guiding principles from the technical appendices supporting Protocol 20 (SABCS 2008, 2011). In brief:

- Each line of evidence was assigned a preliminary "weight" in the problem formulation by considering how closely the line of evidence matched the assessment endpoint (i.e., the desirable ecosystem component being protected). This weighting strongly reflects the attribute group identified by Menzie et al. (1996) as the most highly ranked, as it includes key considerations of biological linkage between measurement endpoint and assessment endpoint, correlation of stressor to response, and utility of measure for judging environmental harm. The "weight" also considered the anticipated uncertainty associated with the line of evidence by reviewing factors such as sensitivity and specificity, study design and data quality objectives, and representativeness.
- Preliminary decision criteria were established for each line of evidence to interpret responses and to align with the goals of the risk assessment. The decision criteria specify when the magnitude of the observed effect is considered acceptable and when the responses are considered indicative of moderate or large environmental effects. Decisions regarding acceptable risk were based on Protocol 1, 20 and the available scientific literature. Each line of evidence was then evaluated against the decision criteria. Decision criteria can be quantitative or qualitative, depending on the circumstances of each line of evidence.
- Distinct but complementary lines of evidence were grouped based on the "source-pathway-receptor" model and consistent with the concept of "evidence groups" as described by Hope and Clarkson (2014). A risk conclusion for each evidence group was made that considered the value or weight of each data set within the evidence group. The aggregation of these data utilized a sequential lines of evidence approach consistent with Hull and Swanson (2006; Figure 4), in which strength, causality, and statistical magnitude of observed response are all considered.
- Specifically, the risk assessment adopted the approach from Hull and Swanson (2006) where COPCs were excluded if their hazard quotients were less than 1. The risk assessment then used a combination of field and laboratory based studies, taking in account the indirect effects associated with physical variables. The aggregate of the data were evaluated in terms of the magnitude of effects as well as the evidence regarding causality. The concept of evidence groups from Hope and Clarkson (2014) was incorporated by developing independent conclusions for each major ecosystem component even if it meant that the same line of evidence was considered twice (i.e., once for each ecosystem component).

Weight-of-evidence assessment does not need to be a quantitative, prescriptive process. Best professional judgement is part of the process even for relatively formal weighting procedures because it provides a balance between the systematic application of decision rules with sufficient flexibility to accommodate professional judgement. To provide transparency, each line of evidence is accompanied by a narrative that describes the site-specific data in the context of the attributes described above; relevant information from the peer-reviewed literature was also used to support the conclusions reached.

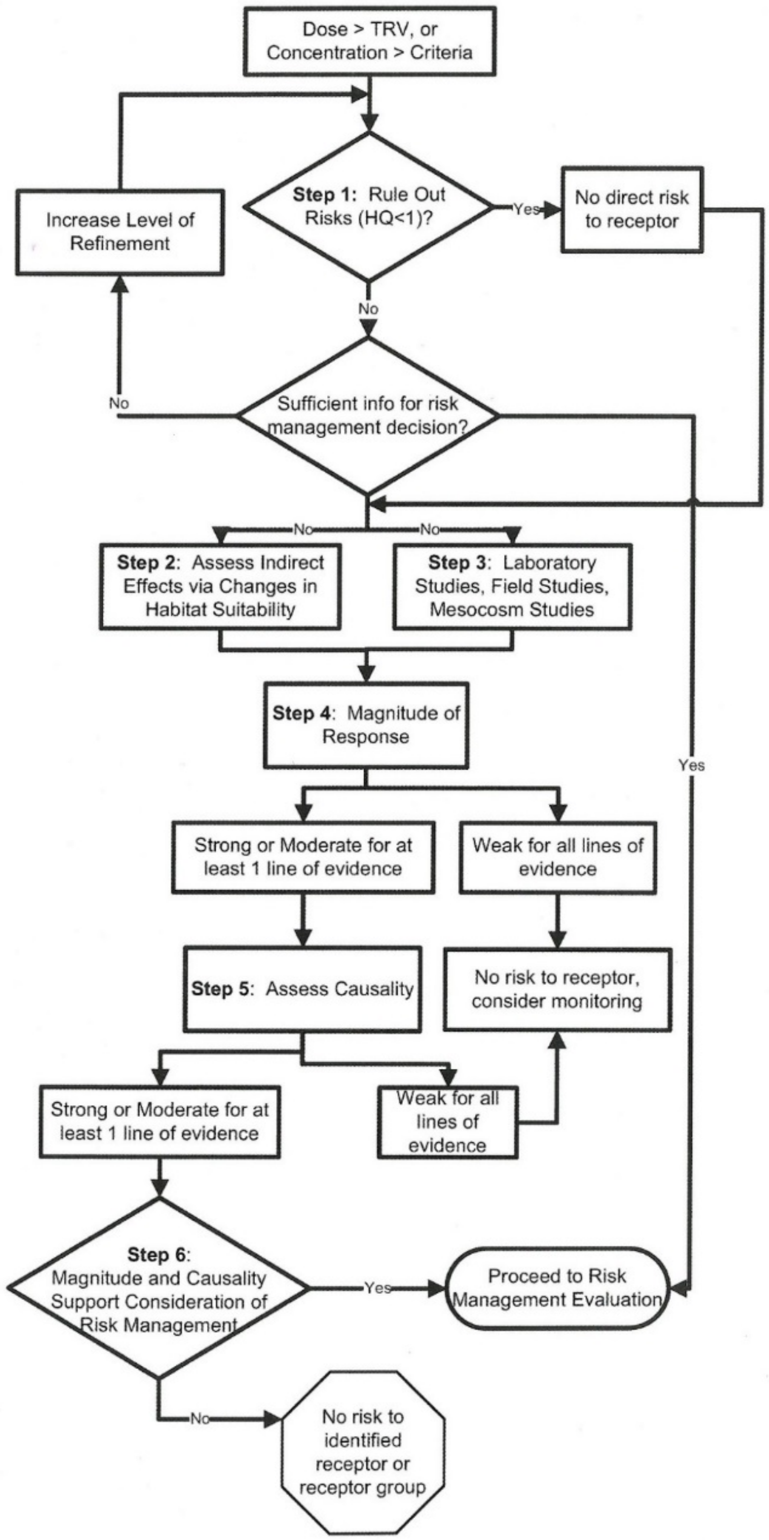

Figure 4: Example of Sequential Lines of Evidence (from Hull and Swanson 2006)

1.3 Report Organization

This current document is organized as follows:

- **Section 1.0 — Introduction**. This section provides regulatory and project context, and describes the overall approach used in the risk assessment.
- **Section 2.0 — Problem formulation**. This section provides a discussion of the objectives and management goals for the ecological risk assessment, followed by the problem formulation steps used to frame how those objectives are achieved. The problem formulation has separate sections for the terrestrial (Section 2.3) and aquatic (Section 2.4) components of the ecological risk assessment, as the tools used for these environments differ. Each section provides:
  - A discussion of the relevant receptors of potential concern and stressors (both contaminant and physical).
  - A graphical representation of the site conceptual model that integrates sources, pathways for stressors, receptor groups, and examines which exposure pathways are relevant to the risk assessment.
  - Identification of assessment endpoints, and a rationale for the selection of specific lines of evidence (also known as measures of effect) for each assessment endpoint.
- **Section 3.0 — Terrestrial weight-of-evidence assessment**. This section provides an overview of methods and findings for lines of evidence related to terrestrial community health. Results are considered in light of the decision criteria established during the problem formulation. The uncertainty around each line of evidence is discussed in terms of its implications for supporting risk management decisions.
- **Section 4.0 — Aquatic weight-of-evidence assessment**. The content is similar to Section 3, but focused on the lines of evidence for the aquatic community.
- **Section 5.0 — Risk Characterization**. This section summarizes the overall risk conclusions, including integration of multiple lines of evidence and implications of the risk conclusions for site management planning (e.g., remediation and long-term monitoring). Opportunities for refinement in risk conclusions in the context of the overall uncertainty are also discussed.

The risk assessment relies on information collected by MPMC and a variety of consultants during the preparation of technical deliverables to address the requirements of the Order and to accommodate information needs by First Nations membership, local residents, and other stakeholders. Much of the technical content has already been presented in the PEEIAR (MPMC 2015), the DSI (Golder 2016b) and a DSI update (Appendix A)[2], and the update to the PEEIAR (Golder 2016a). However, the results of more recent studies are also incorporated. The technical appendices (as presented by the original authors) have been compiled in a single Appendix Book as a companion to the Ecological Risk Assessment report. The body of the Ecological Risk Assessment (this report) provides a brief summary of the study design and sampling methods used in the technical appendix, but then focuses on interpreting the data in the context of a risk assessment. The technical appendices convey further details regarding sampling and analytical methods, quality assurance/quality control procedures, raw data in tabular and graphical form, and the certificates of analysis as presented by the original author. This approach is intended to provide the data used in the risk assessment in a consolidated manner for transparency and for the convenience of reviewers. The Appendix Book is an aggregate of a number of different studies, which, in some cases, have study objectives that were intended to address specific questions that have arisen in MPMC's response to the breach. These studies still contain data relevant to the ecological risk assessment.

[2] Documents that are included as appendices to this risk assessment are described by reference to the appendix letter. Documents that are not included as appendices to the risk assessment are cited as stand-alone documents.

2.0 PROBLEM FORMULATION

2.1 Objective of the Problem Formulation

The elements of the problem formulation follow SABCS (2008) and other risk assessment guidance (Environment Canada 2012a, US EPA 1998). The problem formulation outlines the “why” and “what” for the risk assessment by:

- Defining a management goal that acts as a practical statement about the objectives of the risk assessment with respect to site management planning (SABCS 2008).
- Identifying the major stressors that need to be considered. These include the chemical contaminants of potential concern (COPCs) that are often the focus of a contaminated sites risk assessment, but also physical factors related to, in the present case, landscape erosion and deposition of tailings in the environment. Discriminating between contaminants and physical stressors is important for site management because the cause of observed alterations will influence the selection of management actions and monitoring tools.
- Identifying the receptors of concern. This involves identification of surrogate species or organism groups to represent ecological function, plus consideration of valued ecosystem components that have special regulatory status or particular economic or societal value.
- Identifying the exposure pathways that connect the stressors to the receptors of concern and illustrating those relationships in a conceptual site model.
- Articulating the specific assessment endpoints (attributes to be protected), measures of effect (specific measurements that are considered indicative of ecological response), and decision criteria (rules and procedures for interpreting data) that are used in the weight-of-evidence framework and risk characterization.

The problem formulation helps the risk assessor assemble the available site knowledge and break down a broad environmental question (i.e., is contamination causing adverse effects to the ecosystem?) into a number of manageable parts for which conclusions can be provided and aggregated. It is not possible to investigate all species in the study area, and therefore, identifying the right questions and applying those questions to representative parts of the ecological community helps to add transparency to the risk assessment.

2.2 Definition of Management Goal

The management goal for the terrestrial and aquatic ecological risk assessment is to achieve the continued presence (or successful reintroduction) of a biologically diverse, functional, self-sustaining, and inter-dependent ecosystem. This is the primary goal of ecological risk assessment and risk management as applied to the remediation of contaminated sites in British Columbia as defined by Technical Guidance 7 and this management goal is compatible with the wording used in federal (DFO) objectives of returning habitat productivity in impacted areas. Specific assessment endpoints that lead to achieving a functional and biologically diverse ecosystem are described in Section 2.3.7 for the terrestrial weight-of-evidence framework and in Section 2.4.7 for the aquatic weight-of-evidence framework.

2.3 Terrestrial Problem Formulation

2.3.1 Scenario for Risk Assessment

The terrestrial risk assessment focuses on the terrestrial habitat that was influenced by the deposition of scoured material and tailings as a result of the TSF embankment breach. This study area is referred to as the "Hazeltine corridor" which extends approximately 9 kilometres (km) from the eastern edge of the TSF to the western shoreline of Quesnel Lake. The corridor consists of two different areas:

- The "floodplain" area is where the forest and portions of native soils were removed by the flood of water and debris.
- The "halo" area is where the forest floor was relatively undisturbed, but was buried to varying depths by the deposits of scoured material and tailings.

MPMC have been implementing rehabilitation and remediation works since the breach and have removed tailings from much of the upper and lower portions of the corridor. The terrestrial ecological risk assessment has incorporated the post-rehabilitation conditions to the extent possible, primarily in terms of the food chain modelling (Section 3.4.2). Conceptually, the physical habitat alteration from scouring and deposition was the dominant impact mechanism. The rehabilitation program is focused on enhancing the natural processes (e.g., deposition of additional organic material; hydraulic connectivity; gradual recolonization by soil organisms) which will gradually reintroduce a biologically diverse, functional, self-sustaining, and inter-dependent ecosystem. The design of the terrestrial risk assessment is explicitly focused on determining whether the contaminant stressor (i.e., residual concentrations of metals from tailings) represent an unacceptable risk that requires risk management actions over and above the rehabilitation and monitored natural recovery already underway.

2.3.2 Selection of Contaminants of Potential Concern (COPCs)

2.3.2.1 Screening Process

The general process for evaluating COPCs for the terrestrial ecological risk assessment included a conservative starting point (preliminary screening) which was then narrowed down to focus the risk assessment on the specific substances that have the potential to present a toxicological hazard. This process involved:

- All COPCs identified in the DSI were considered as a preliminary contaminant of potential concern. The available soil, surface water, and groundwater chemistry data from the DSI were compared to the applicable numerical standards. A substance was retained at this stage if measured concentrations exceeded the standard.
- The preliminary COPCs were subsequently narrowed down by evaluating whether the contamination was present over large areas of the site. For soil, this was accomplished by comparing a reasonable worst-case exposure concentration (e.g., 95% upper confidence limit of the mean [UCLM] or 90th percentile) to the contaminated sites regulation (CSR) parkland use standards for the protection of soil invertebrates and plants, and also to the agricultural land use standards for livestock ingesting soil and fodder. The agricultural land use standards were applied because a small area near Hazeltine corridor is contained under the Agricultural Land Reserve.

- The preliminary list was further refined by considering the local background concentrations. Preliminary COPCs were excluded if the maximum observed concentration from the area of interest did not exceed the background concentration (i.e., 95th percentile).

2.3.2.2 Review of the DSI Soil Chemistry With Respect to Terrestrial COPCs

The detailed site investigation (Golder 2016b, Appendix A) was the primary source of soil chemistry data used in the ecological risk assessment. Table 1 provides a summary of the COPCs based on the soil chemistry data (Appendix A) available from the following sources:

- The initial soil quality impact assessment was completed by SNC-Lavalin as part of the PEEIAR (MPMC 2015)—This data set consisted of 71 samples from 68 stations collected between September to October 2014 (i.e., within three months of the TSF embankment breach) from visually identified impacted areas of Hazeltine corridor. A further 23 samples were collected from 19 stations in local background areas (i.e., undisturbed areas beyond the extent of visual impact). The design for this sampling program entailed collection of samples across 18 transects spaced at approximately 500-metre (m) intervals along the corridor. Transects extended outside the area of influence to include native soils. Across the width of each transect, samples were collected from deposited tailings as well as native soil below tailings[3].
- The soil sampling completed as part of the DSI (Golder 2016b; Appendix A)—This data set added 55 samples collected from 34 stations within the Hazeltine corridor, plus 56 samples from 45 stations in local background areas. Samples were collected in August 2015 and August 2016 for two purposes: First, samples were collected for comparison to the spatial profiles described in the PEEIAR study; samples were collected at seven transects previously established by SNC-Lavalin and followed the same approach as the original data set described above. Second, samples were collected as co-located samples with plant and soil invertebrate tissue samples to provide a synoptic basis for evaluating bioaccumulation of metals. For these samples, typically the top 0.2 m of soil was sampled to represent the most biologically active zone.
- The data report on local background soil quality, completed as part of the DSI (Appendix A-1)—This data set consisted of all background data collected during the 2014, 2015, and 2016 programs described above, plus an additional 12 samples collected from four stations within local background areas. Local background data were collected in accordance with requirements documented in CSR Protocol 4 and Technical Guidance 16. The data were not submitted for review to the BC MoE as the local background concentrations were not used as an alternative to a standard or regional background concentrations outlined in Protocol 4.
- A total of 182 soil samples that were classified as tailings-influenced (i.e., they were not native soils present in the Hazeltine corridor prior the breach) were available.

[3] See Section 3 of MPMC (2015) Appendix D for a discussion of sampling and analytical methods. The differentiation between “tailings” and “native soil” samples was also made in the DSI (Golder 2016a). From a risk assessment perspective, the tailings samples provide the most realistic data set for identifying contaminants of potential concern related to the breach.

Table 1: Summary of Preliminary Soil Contaminants of Potential Concern at Completion of the DSI

Contaminant	Applicable CSR Standard	CSR Schedule	Maximum	90th Percentile	95% UCLM[a]
Copper	150	5 – Toxicity to soil invertebrates and plants	1,560	1,000	737
Sulphur	500	4 – Generic, Agricultural	3,500	1,550	1,615
Molybdenum	5	4 – Generic, Agricultural	7.3	5.4	4.0
Vanadium	200	4 – Generic, Agricultural and Parkland	289	213	177

a) CSR = Contaminated Sites Regulation; DSI = Detailed Site Investigation; UCLM = upper confidence limit of the mean. Data are from DSI, Table 6.

Several other substances in excess of numerical standards were present in a small proportion (<1.1%) of samples, but these were excluded as COPCs in the DSI because the maximum concentrations did not exceed the standard by a factor greater than 2, and because the calculated 90th percentile and 95% UCLM did not exceed the standard. The use of statistics to characterize soil quality (and thus, to inform the selection of COPCs for the ecological risk assessment) was conducted in accordance with Technical Guidance 2 (Appendix A). Substances that were excluded from further consideration in soil based on this statistical approach included arsenic, barium, and cadmium. The number of samples that exceeded standards for these three excluded substances was one or two samples of a total of 182 samples. Further details can be found in Table 4 of Appendix A.

2.3.2.3 Further Refinement of Soil COPCs Based on Toxicological Hazards

The purpose of the data screening described above was to identify those substances that have the potential to present unacceptable toxicological hazards to ecological receptors. Copper is relatively well understood in terms of its toxicological hazard, and there are CSR matrix standards available to evaluate its potential hazard to soil invertebrates and plants, as well as US EPA ecological soil screening level values[4] to evaluate potential hazard to soil invertebrates, plants, mammals, and birds. Vanadium does not have a matrix-specific numerical standard under the CSR (it has a generic standard), but does have ecological soil screening level values to evaluate hazards to mammals and birds, as well as a Canadian Council of Ministers of the Environment (CCME) guideline for the protection of soil invertebrates and plants (CCME 1997). Conversely, the available scientific literature is less clear with respect to ecological risk of molybdenum and sulphur exposures. These substances were reviewed in further detail to determine if they should be retained in the ecological risk assessment:

[4] https://www.epa.gov/risk/ecological-soil-screening-level-eco-ssl-guidance-and-documents.

- Molybdenum—Molybdenum was excluded from further consideration as a contaminant of potential concern (COPC) because it was associated with natural conditions and not the deposition of tailings (Appendix A). Molybdenum was included in the DSI stage of investigation because of the small area where the agricultural land standards may potentially apply; the molybdenum generic numerical soil standard is 5 mg/kg for agricultural land use. However, molybdenum can be excluded using the statistical approach described by Technical Guidance 2 in a manner similar to arsenic, barium, and cadmium. Further consideration of the potential toxicological hazards of molybdenum was conducted to strengthen the confidence in this approach. The provincial standards were adopted from CCME (1999a) which provided interim guidelines for use at contaminated sites. These interim guidelines were based on professional judgement to assist managers of contaminated sites while soil quality guidelines were derived (CCME 1999b). A detailed review of the available toxicological data following the CCME derivation method (CCME 1999c) was subsequently completed (Milllenium 2014) that resulted in a proposed value of 32 mg/kg for the protection of soil invertebrates and plants for natural areas, residential and parkland sites. A value of 50 mg/kg was proposed for protection of mammals in natural areas[5]. Molybdenum is also an essential micronutrient for animals (Rajagopalan 1988) and plays an important role in nitrogen metabolism for plants (Jones 1994). High levels of molybdenum in plant tissue can result in molybdenum-induced copper deficiency (molybdenosis) in ruminants (Allen and Gawthorne 1987; Owen 1999). The occurrence and severity of molybdenosis is inversely proportional to dietary copper; copper to molybdenum ratios of 2:1 or greater minimizes the risk of molybdenosis (Jones 1994; Owen 1999). The ratios of plant tissue concentrations of copper to molybdenum were well above 2:1 based on the data collected from the site. Golder concludes that molybdenum can be eliminated from the quantitative risk analysis because the maximum observed soil concentration in tailings is well below the available toxicologically-based screening values. There is no evidence that an imbalance in metal uptake by plants would introduce an additional hazard to herbivores.
- Sulphur—Sulphur was excluded from further consideration as a COPC because it was associated with natural conditions and not the deposition of tailings (Appendix A). Local background concentrations of 900 mg/kg and 600 mg/kg were derived for organic and mineral soils, respectively (Table 3, Appendix A); both of these values exceeded the soil standard of 500 mg/kg. The DSI concluded that sulphur is present from natural sources, and is already elevated in the background samples. Furthermore, the CSR standard (and the CCME interim soil quality guideline[6]) is based on elemental sulphur present in a sample. The available soil chemistry data are measurements of total sulphur (not the fraction that exists as elemental sulphur); therefore screening to the CSR standard is conservative (i.e., cautionary). Sulphur is not known to be toxic when ingested except in very high doses to ruminants such as in concentrations greater than 0.3 to 0.4% (i.e., equivalent to 3,000 mg/kg to 4,000 mg/kg of sulphur) as sulphate or elemental sulphur in foods (Kandylis 1984). Sulphur can therefore be excluded from the risk assessment because the analytical method for total sulphur overestimates the amount of sulphur present in the elemental form that is the basis of the standard and guideline. Sulphur is also an essential mineral that is necessary for biological function, as it is a component of DNA, amino acids, vitamins and insulin (Underwood and Suttle 1999).

[5] Soil concentration is back calculated from the wildlife toxicity reference value from Oak Ridge National Laboratory using default assumptions for a vole.

[6] The soil quality guideline from CCME for sulphur is also based on professional judgement and was intended as an interim value pending the derivation of a toxicologically-relevant soil quality guideline.

In summary, molybdenum and sulphur were excluded as COPCs in soil based on the hazard assessment and ecological relevance check conducted as part of the problem formulation. Copper and vanadium were retained for further evaluation.

2.3.2.4 Review of the DSI Water Chemistry with Respect to Terrestrial COPCs

Surface water chemistry data were reviewed in terms of potential hazards to mammals and birds that could consume drinking water from waterbodies impacted by the TSF embankment breach (i.e., Hazeltine Channel, Quesnel Lake, and Polley Lake). This review focused on monitoring data collected between May and November 2015 that were reported in the PEEIAR and the DSI update. This time frame is after the completion of majority of in-stream works in Hazeltine Channel but before the discharge of mine effluent. The primary basis for COPC selection was comparison of the water chemistry data to the lowest of the CSR Generic Numerical Water Standards for Livestock Water or the Canadian Water Quality Guidelines for the Protection of Agriculture Water Uses – Livestock. For parameters lacking these standards or guidelines, either the CSR Schedule 6 Generic Numerical Water Standards or Schedule 10 Generic Numerical Soil and Water Standards for Drinking Water was applied. The screening was two-tiered:

- A substance was considered a preliminary COPC if its maximum concentration exceeded the applicable screening value described above.
- The preliminary COPCs were narrowed down by evaluating whether or not the substance was present at elevated concentration over a chronic exposure duration. The evaluation compared the 95th percentile concentration to the applicable screening value described above.

Substances that did not have a livestock-based standard or guideline (or a generic CSR water standard) were not evaluated. Groundwater was not screened for the terrestrial risk assessment because wildlife do not consume groundwater. Shallow groundwater flow at the site was inferred to discharge to Hazeltine Channel and the migration of groundwater contamination is low based on the results of the geochemical characterization, kinetic leach testing and groundwater analytical data (Golder 2016a, Section 6.1.4). Surface water data included samples from the Hazeltine Channel system to capture the potential exposure of groundwater flowing to waterbodies consumed by wildlife as drinking water. There were no chemical contaminants of potential concern identified in terms of wildlife consuming surface water (Table 2).

Table 2: Comparison of Surface Water Concentrations to Wildlife Screening Values

Parameter	Applicable Standard or Guideline (mg/L)	CSR Schedule or BC WQG Matrix	Summary Concentration	
			Maximum	95th Percentile
Aluminum	5	Schedule 6—Livestock Drinking Water	**6.4**	2.01
Copper (total)	0.3	Schedule 6—Livestock Drinking Water	**0.47**	0.06
Iron	6.5	Schedule 6—Generic Drinking Water	**8.7**	2.5

Data are from the Hazeltine Channel between May and November 2015. Bold underlined values indicate the concentration was greater than the screening value.

CSR = Contaminated Sites Regulation; WQG = water quality guideline.

2.3.2.5 Final Selection of COPCs for the Terrestrial Risk Assessment

At the completion of the screening process, copper and vanadium in soil were identified as COPCs for the terrestrial ecological risk assessment[7]. Identification of copper and vanadium as the priority substances of interest for evaluation in the risk assessment does not mean that other metals or metalloids are not being evaluated. Many of the lines of evidence involve measurement of biological or ecological effects as a result of exposure to the mixture of metals and metalloids that is present in tailings. Focusing on priority substances such as copper and vanadium is appropriate and consistent with risk assessment practice, particularly when evaluating substances that have toxicologically-based screening values that have been adopted as part of provincial regulation regarding the management of contaminated sites. Often, substances lack a numerical guideline, standard, or screening value because they are generally not of toxicological interest or rarely encountered at contaminated sites.

2.3.3 Terrestrial Receptors of Potential Concern

This section highlights receptors of potential concern for the terrestrial risk assessment. The risk assessment focuses on representative species selected from different groups of wildlife species (e.g., small mammal herbivores) based on consideration of the site's ecology. The risk characterization is ultimately based on a number of lines of evidence ("measures of effect") that reflect specific assessment endpoints ("what ecological attributes are being protected") which in turn are linked to these specific species or ecosystem components. Information regarding the overall ecology and observed plant and wildlife species in the terrestrial study area was collected as part of the PEEIAR. A technical data report was prepared by SNC-Lavalin (MPMC 2015) which was used as the primary basis for selecting terrestrial receptors of concern, supplemented by observations by field staff and site personnel where appropriate.

2.3.3.1 Terrestrial Vegetation

A detailed description of the terrestrial vegetation in the study area is provided in MPMC (2015). Additional plant inventories in the background forest and Hazeltine corridor areas were conducted by Golder and MPMC in the summer of 2015 and 2016 (Appendix H-1; Appendix H-2). A detailed listing of specific species inventoried, and observed changes over time is provided in those appendices. In brief, the plant community is composed of species consistent with the biogeoclimatic zone (Interior Cedar Hemlock) and its related variants throughout the site. Areas impacted by the TSF embankment breach have limited vegetation which is now gradually being re-established, both through natural ingress and through planting. MPMC have also re-planted a variety of native riparian and upper bank trees, shrubs, forbes and herbs. Examples of the major plant types that are currently present in the disturbed areas include:

- Grasses—Rye and barley grass *(Lolium* sp., *Secale* sp., and *Hordeum* sp.) are present in the floodplain, in addition to fescue (*Festuca)*, meadow grass (*Poa*), woodreed (*Cinna*), tufted hairgrass (*Deschampsia*) and needle grass (*Achnatherum*).
- Shrubs—Shrubs include thimbleberry (*Rubus parviflorus*), blueberry (*Vaccinium* sp.), Devil's club (*Oplopanax horridus*), and highbush cranberry (*Viburnum trilobum*).

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[7] Although these substances were identified in soil, the use of a mechanistic food chain model also means that risks to wildlife from copper and vanadium are evaluated in all media.

- Trees—The halo contains mature trees and a limited understory consisting of a variety of tree and shrub species. The terrestrial study area is surrounded by a dominantly coniferous forest type which consists of a seral shrub and young deciduous forest – dominated by trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), green alder (*Alnus viridis*), and willow (*Salix sp.)* as well as a mature conifer forest – dominated by lodgepole pine (*Pinus contorta*) and white spruce (*Picea glauca).*

2.3.3.2 Terrestrial Wildlife

Wildlife species observed in the vicinity of the Mine during terrestrial assessment fieldwork conducted after the TSF embankment breach (MPMC 2015; Appendix H-1; Appendix H-2) are listed in Table 3. Table 3 also includes wildlife species observed by MPMC staff in 2014 and 2015 (Shauna Litke, personal communication, 19 January 2016; Colleen Hughes, personal communication, 20 January, 2016). This list is not a comprehensive list of resident species but likely represents the more common species that use the area throughout the year. A list of bird species expected to occur in the Cariboo region was obtained from the BC Bird Atlas and is presented in the inventory prepared by SNC-Lavalin (MPMC 2015). The list includes bird species present in the entire region and not all of them will be found in the vicinity of terrestrial study area.

Table 3: Wildlife Species Detected in the Vicinity of the Mine, 2014 to 2016

Reptiles and Amphibians			
Common gartersnake	Pacific treefrog	Western toad	Columbia spotted frog
Birds			
Mallard Bufflehead Ring-necked duck Common goldeneye Northern pygmy-owl Hairy woodpecker Pileated woodpecker	Ruffed grouse Spruce grouse Bald eagle Osprey Golden eagle Killdeer Great blue heron	Sandhill crane Hooded merganser Common merganser Common loon Steller's jay Belted kingfisher American dipper	Northern harrier Red-tailed hawk Canada goose Trumpeter swan Hummingbird Chickadee Northern flicker
Mammals			
Moose Mule deer Beaver Gray wolf Grizzly bear Black bear	Cougar Lynx Red fox River otter Red squirrel	Snowshoe hare Coyote Bobcat Skunk Muskat	Packrat Ermine Pine Marten Fisher Wolverine

Source: Appendix H; also Shauna Litke, personal communications, 19 January 2016 and Colleen Hughes, personal communication, 20 January 2016.

2.3.3.3 Identification of Listed and Valued Terrestrial Species

Species that have been listed as endangered or at risk by provincial or federal authorities, or that have particular economic or societal value are typically emphasized in the selection of receptors of potential concern.

There were 72 blue or red-listed[8] plants that had the potential to occur in the general vicinity of the Mine, based on a review of rare plant habitat affinities, the ecosystems that likely occurred in the terrestrial study area, and a survey of the BC online database (MPMC 2015).

Table 4 summarizes at-risk terrestrial wildlife with the potential to occur because potential habitat was present in the vicinity of the terrestrial study area (MPMC 2015). Grizzly bears (*Ursus arctos*) are not considered to be at risk in this specific study area because they are not managed by the province in the area west of the West Arm of Quesnel Lake. Grizzly bears were recorded during the 2016 terrestrial ecosystem assessment surveys (Appendix H-1). The great blue heron (*Ardea herodias*) was listed as a blue listed species (MPMC 2015), however, only the coastal sub species (*Ardea herodias fannini*) is considered to be of special concern, and individuals from that population would not be present within the vicinity of the Mount Polley mine. Western toad (*Anaxyrus boreas*) has been observed to be widely distributed in the study area both before and after the TSF embankment breach (Appendix G-2). In addition to listed species, provincial authorities also manage a number of species of regional importance in the broader Cariboo region, including moose, mule deer, and bald eagle. Other species may have economic or societal importance in terms of trapping or hunting (e.g., mink, weasel, marten, otter, coyote, fox, squirrel, muskrat, beaver and black bear). Other species such as waterfowl and game birds (e.g., ruffed grouse or spruce grouse) may also be hunted.

Table 4: At-Risk Wildlife Species Potentially Present in the Vicinity of the Terrestrial Study Area.

Wildlife Species	At-Risk Listing
Insects	
Hagen's bluet	Blue (BC)
Magnum mantleslug	Blue (BC)
Reptiles and Amphibians	
Western toad	Blue (BC); Special Concern (SARA)
Birds	
Common nighthawk	Threatened (SARA)
Olive-sided flycatcher	Blue (BC); Threatened (SARA)
Mammals	
Little brown myotis	Endangered (SARA)
Northern myotis	Blue (BC); Endangered (SARA)
Fisher	Blue (BC)
Wolverine	Blue (BC)
Grizzly bear	Blue (BC)

"Blue (BC)"—species considered to be of "special concern" in BC.

"Special Concern (SARA)"—species determined by the *Species at Risk Act (SARA)* to be of special concern because of characteristics that make it is particularly sensitive to human activities or natural events.

"Threatened (SARA)"—species determined by the Species at Risk Act that is likely to become endangered if limiting factors are not reversed.

"Endangered (SARA)"—species facing imminent extirpation or extinction.

[8] Red (BC) is a species considered to be endangered or threatened by provincial authorities. Blue (BC) is a species considered to be of "special concern" by provincial authorities.

2.3.3.4 Receptors of Potential Concern

Table 5 lists the receptor groups for the terrestrial risk assessment (along with example taxonomic groups or species) that were selected based on the available site information.

Table 5: Receptor Groups with Representative Taxonomic Group or Species for Terrestrial Risk Assessment

Receptor Group	Feeding Guild	Example Taxonomic Group or Species
Lower Trophic Level		
Soil Microbes	Decomposers	Bacteria, fungi
Terrestrial Plants	Primary producers	Moss, grass, shrubs, trees
Soil Invertebrates	Primary & secondary consumers	Earthworms, ants, beetles, spiders
Higher Trophic Level		
Herpetofauna	Insectivorous	Western toad
	Carnivorous	Common gartersnake
Birds	Herbivorous	Dark-eyed junco
	Insectivorous	Robin, common nighthawk, olive-sided flycatcher
	Omnivorous	Common raven
	Carnivorous	Northern pygmy-owl
	Piscivorous	Osprey, great blue heron
Mammals	Herbivorous	Meadow vole, Mule deer
	Insectivorous	Masked shrew
	Omnivorous	Deer mouse
	Carnivorous	Ermine, fisher
	Piscivorous	River otter

2.3.4 Identification of Terrestrial Contaminant Exposure Pathways

There are several routes by which a terrestrial organism can be exposed to a given contaminant. The route of exposure often influences both the site of toxic action and toxicity mechanism for the organism (McGeer et al. 2003). There is also the physical effect associated with the initial scouring and deposition of material during the TSF embankment breach which is an additional stressor that needs to be considered in the interpretation of each individual line of evidence. Terrestrial receptors described in Table 5 could be exposed to the COPCs via a combination of one or more of the following pathways:

- Direct exposure to soil
- Ingestion of soil
- Ingestion of water
- Ingestion of food items

These four main exposure pathways are described below for each of the major receptor groups in the context of how the environmental fate, transport and bioavailability of copper (and to a lesser degree, vanadium) under site-specific conditions will likely influence the relative contribution of each operable pathway to risks for a given receptor. This discussion is meant to highlight potential factors that may influence the relative magnitude of the different operable exposure pathways. An appreciation of these important factors helps improve the realism of the problem formulation, and by extension, improves the linkage between the selected lines of evidence and the overall assessment endpoint.

Uptake by Microbes

Soil microbes are exposed to soluble copper in soil. Microbial activity is key to the symbiotic nutrient-sharing network between the soil microbial community and forest vegetation (Simard et al. 1997). Mycorrhizal fungi provide plants with nutrients in exchange for energy generated through photosynthesis. Microbes are also key to the gradual decomposition of organic material and the gradual re-establishment of an organic soil layer in the areas impacted by tailings.

Uptake by Plants

Plants are exposed to COPCs via transdermal exposure to soluble metals in soil porewater or in groundwater. They use multiple strategies for tolerating excess metals in soils, including restriction of uptake at the root surface (e.g., through cellular regulation, excluding substances that bind metal, manipulating rhizosphere pH), restriction of transport through plant tissues, and through internal tolerance mechanisms (e.g., immobilization, exclusion, chelation with peptide ligands, compartmentalization of the metal ions in vacuoles).

Uptake by Soil Dwelling Invertebrates

Soil invertebrates are exposed to COPCs via soluble metals in soil and by ingesting soil and prey and passing it through the digestive tract to extract nutrients. The character of foraging substrate and the food sources present also influence which taxonomic groups of soil organisms will be dominant (Hawkins and MacMahon 1989). Soil microbes and macroinvertebrates play an important role in nutrient-sharing and decomposition of organic matter in terrestrial systems. The life histories of soil invertebrates are diverse, with different organisms adapted to exploit different types of physical habitat and food sources. Typical feeding guilds include “herbivores” (that process plant matter), “detritivores” (that process dead and decaying plant or animal matter), and “predators” (that feed on other invertebrates).

Uptake by Amphibians

Amphibians are exposed in a manner similar to reptiles, mammals and birds when occupying terrestrial habitats (e.g., they ingest incidental soil as well as prey items), but are exposed in a manner similar to fish and aquatic invertebrates when utilizing aquatic habitats (e.g., direct contact with water; see Section 2.4.4 of the aquatic problem formulation).

Uptake by Reptiles, Mammals and Birds

Mammals, birds, and reptiles are exposed to COPCs via ingestion of food items, the incidental ingestion of soil during feeding, and the ingestion of drinking water. Metals that have bioaccumulated in the tissues of terrestrial plants, soil invertebrates, fish or amphibians are transferred to wildlife receptors depending on their dietary preferences. Copper and vanadium can be transferred among trophic levels as each successive predator in the food web ingests various prey items. The cumulative ingested dose (i.e., the sum of food, soil and water) is then subject to gastrointestinal processes that influence the fraction of the ingested dose that is actually absorbed, and a large proportion of the ingested dose is ultimately excreted. This absorbed fraction is then partitioned into specific tissues within the organism depending on a number of factors, including internal sequestration, storage, elimination kinetics, and nutrient essentiality. A fraction of the absorbed dose is then available to exert toxicity on specific organs and cellular structures. There is no evidence that copper or vanadium biomagnify in terrestrial systems (i.e., concentrations increase with trophic level).

2.3.5 Physical Influences of Tailings and Debris

The physical effect of the tailings release on terrestrial organisms and their habitat is an important consideration in the terrestrial risk assessment. As noted in Section 2.3.1, the focus of the terrestrial risk assessment is to understand whether adverse effects related to contamination are present over and above the physical alterations (for which remediation and monitored natural recovery activities are already underway). Many of the lines of evidence used to evaluate those risks will be influenced to some degree by the physical stressor. Specific examples of these confounding effects include:

- Natural soil consists of particles of varying sizes (e.g., from silts to gravels) with substantial amounts of organic carbon. Conversely, the deposited tailings consist of finely-ground rock with little to no organic carbon. Tailings are unlikely to have the same ability to hold soil moisture and to provide suitable food or substrates for plants or soil invertebrates. Soil toxicity testing may demonstrate a reduced growth or survival in plants that is due to these physical factors rather than toxicological influences of soil contamination.
- Areas where tailings overlay native soils could impede air exchange with plant roots and the soil microbial community, and thus alter biogeochemical cycling. Adverse effects on the diversity or abundance of microbial, plant or soil invertebrate communities may be present as a result of these physical impacts that are unrelated to contaminants.
- The physical alteration from scouring and deposition has resulted in reduced habitat variability, quality and quantity in the impacted areas. This could alter the availability of prey items and habitat usage by wildlife species. This change could result in either increased or decreased use by different wildlife receptors, depending on the successional stage during recovery and the abundance and diversity of preferred prey items.
- Ultimately, physical impacts related to the deposition of tailings in the terrestrial environment are being addressed through ongoing rehabilitation efforts. The risk assessment provides information about the relative influence of the physical stressor versus the potential for contaminant-related risks to help refine and inform the ongoing risk management actions already underway.

2.3.6 Terrestrial Conceptual Site Model

The conceptual site model (Figure 5) integrates the source of COPCs, operable exposure pathways, and receptors of potential concern into a single figure. The purpose of the conceptual site model is to distill the available information into a more simplified depiction of the key risk pathways to help convey the associated hypotheses concerning the nature of ecological risks at a contaminated site.

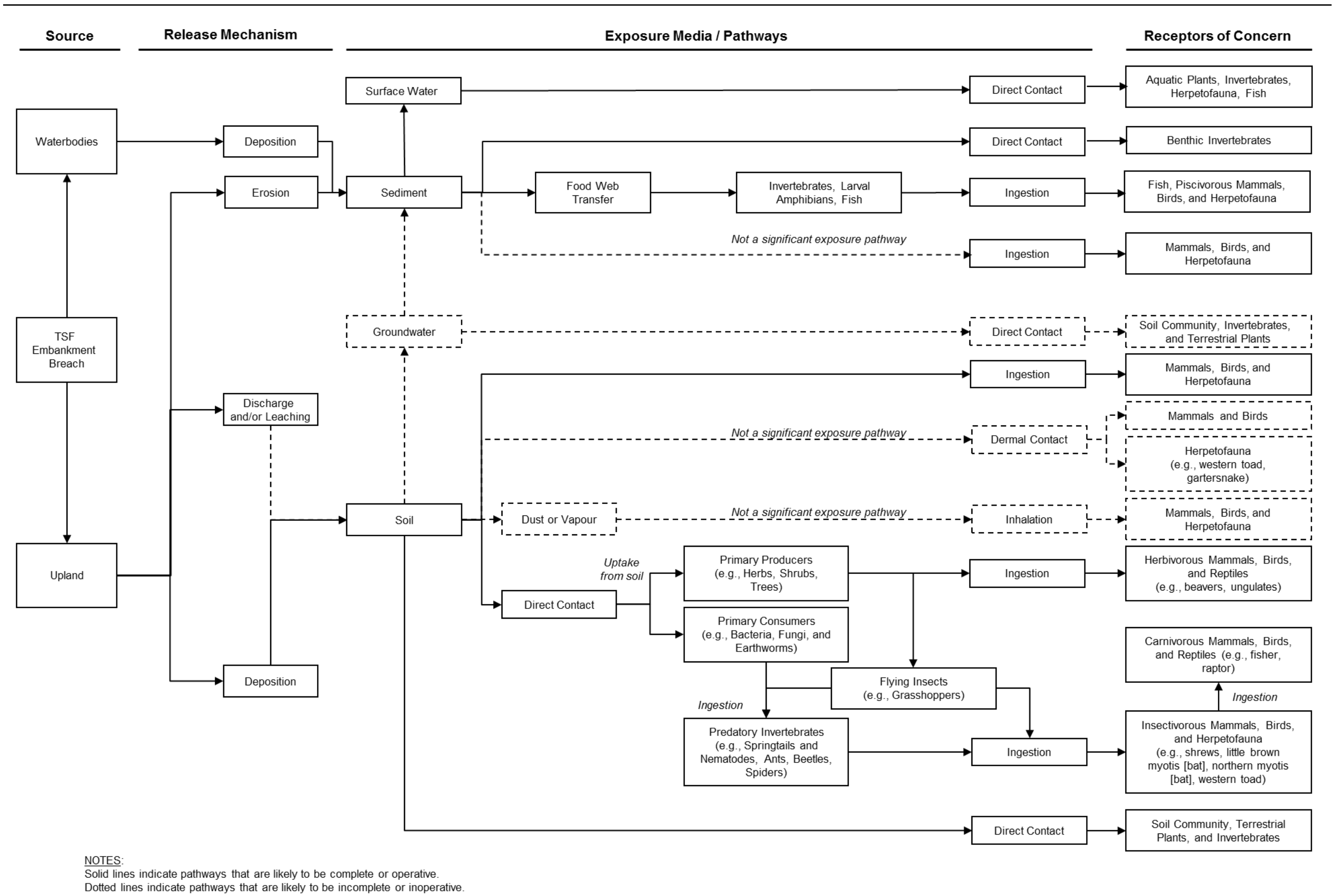

Figure 5: Conceptual Site Model for Ecological Risk Assessment

2.3.7 Application of the Weight-of-Evidence Framework

The weight-of-evidence framework was introduced in Section 1.2.2. The weight-of-evidence framework helps to provide clarity about what types of information are available for the risk assessment, and how such information will be evaluated. The framework also helps to determine how much "weight" or strength a particular line of evidence will be assigned in the overall risk conclusion. The terrestrial risk assessment considered the following assessment endpoints and major lines of evidence:

- Assessment Endpoint 1: Protect microbes, soil invertebrates and plants from potentially hazardous magnitude of exposure from metals via direct contact with soil. This assessment endpoint focuses on understanding the potential hazards associated with total concentrations of the COPCs in the environment. It corresponds to the "exposure assessment" in the classic risk assessment paradigm. Lines of evidence related to this assessment endpoint focus on comparing the total concentrations of COPCs to risk-based numerical guidelines, standards or screening values, as well as surrogate measurements of how much of the total concentration is likely be mobile or bioavailable (e.g., leaching potential from tailings). Although chemical exposures to biota are included as an assessment endpoint, the importance of this endpoint is lower than for the effects-based measures described below, which are more closely associated with the functional ecological values of interest.
- Assessment Endpoint 2: Protect soil invertebrates and plants (collectively, the lower trophic levels) from reductions in survival, growth and reproduction as a result of metals in tailings. This assessment endpoint focuses on lines of evidence that measure effects (either in the field or in the laboratory), or that provide a direct evaluation of bioavailability. Examples of lines of evidence considered for this assessment endpoint included a quantitative forest health assessment, measurement of adverse effects in the laboratory using standardized toxicity tests; and comparison of the concentrations of metals in invertebrates and plants between exposed and reference areas.
- Assessment Endpoint 3: Protect terrestrial wildlife from reduction in survival, growth and reproduction as a result of exposure via ingestion of food items and incidental soil. A variety of relevant receptor species from different feeding guilds (collectively, the higher trophic levels encompassing birds and mammals) was evaluated using a food chain model that compared the estimated daily ingested dose to the toxicologically "safe" concentration from the literature.

The specific lines of evidence for each of these major assessment endpoints are described in further detail in Section 3.0.

2.4 Aquatic Problem Formulation

2.4.1 Scenario for Risk Assessment

The aquatic risk assessment focuses on the aquatic habitats that were influenced by the deposition of scoured material and tailings as a result of the TSF embankment breach. Such material was deposited into both Polley and Quesnel Lakes, along the Hazeltine Channel, and at the mouth of Edney Creek. Displaced material and tailings entered Quesnel and Polley Lakes. MPMC has been implementing a rehabilitation strategy over the last two years that resulted in the removal of tailings from the Hazeltine Channel bed and surrounding areas, reconstruction of the Hazeltine and Edney Creek channels, and restoration of the connection of Edney Creek with Quesnel Lake. The design of the aquatic risk assessment is explicitly focused on determining whether the contaminant stressors (i.e., residual concentrations of metals and metalloids from tailings) represent an unacceptable risk that requires remediation over and above the reconstruction and monitored natural recovery already underway.

2.4.2 Identification of Contaminants of Potential Concern (COPCs)

2.4.2.1 General Process

The intent of screening data to select COPCs was to help focus the risk assessment on the specific substances that are associated with the breach and are present in the environment at concentrations that have the potential to exert a negative effect on the receptors of potential concern. The process for identifying COPCs for the ecological risk assessment consisted of:

- A preliminary list of COPCs was established by comparing the concentrations of the substances to: a) applicable provincial ambient guidelines, or b) in the absence of a provincial standard or guideline, a risk-based screening value from a different jurisdiction.
- The preliminary list was refined by considering background concentrations. The purposes of this refinement step is to keep the risk assessment focused on those substances that are directly related to tailings and also have the highest potential to cause harm to aquatic life. Substances that were added as preliminary COPCs were removed if their reasonable worst-case concentration in the area of interest did not exceed the background concentration.

2.4.2.2 Review of the DSI Surface Water Chemistry With Respect to Aquatic COPCs

Surface water chemistry data were reviewed in terms of potential hazards to aquatic organism that could consume drinking water from waterbodies impacted by the TSF embankment breach. This review focused on monitoring data collected between May and November 2015 that were reported in the PEEIAR (MPMC 2015) and the DSI update (Appendix A). This time frame provides a conservative selection of COPCs based on the potential long-term exposure to substances in water without being influenced by the short-term conditions immediately after the breach. Specifically:

- Data from Quesnel and Polley Lakes prior to March 2015 were influenced by suspended particulates from the original release of tailings, which typically settled out of the water column through late 2014 and into early 2015. Data collected after March 2015 is likely representative of the future conditions.
- Data from the Hazeltine Channel between May and November 2015 is after the completion of majority of in-stream works but before the discharge of mine effluent.

The primary basis for COPC selection was comparison of the water chemistry data to the BC approved ambient water quality guidelines (WQG) for the protection of aquatic life (Table 6). Working WQGs for the protection of aquatic life were also considered. The screening consisted of first checking if the maximum concentrations for each receiving environment area exceeded the guideline values. The 95th percentile concentration was calculated for each substance that had a maximum concentration greater than applicable guidelines. The substance was retained as a COPC if the 95th percentile concentration from the receiving area was greater than the applicable guidelines and the pre-breach background concentration[9]. Substances that did not have an approved or working WQGs were not considered as potential COPCs. This approach was used in the initial water quality assessment (MPMC 2015) which concluded that substances without a working or approved water quality guideline were unlikely to be of environmental concern.

Table 6. Metals Exceedances based on Maximum and 95th Percentile Concentrations between 1 May and 30 November 2015

	BC WQG[a]		Polley Lake[b]		Hazeltine Channel[b]		Quesnel Lake[b]	
	Maximum	30-day Average	Maximum	95th Percentile	Maximum	95th Percentile	Maximum	95th Percentile
Total Aluminum	-	5	0.2	0.066	<u>6.4</u>	2.0	0.22	0.077
Total Arsenic	-	0.005	0.0011	0.001	<u>0.0098</u>	0.0027	0.00034	0.00018
Total Beryllium	-	0.00013	0.00005	0.00005	<u>0.00029</u>	0.00005	0.00005	0.00005
Total Chromium	-	0.001	0.00025	0.00025	<u>0.0054</u>	<u>0.0028</u>	0.00025	0.00025
Total Cobalt	0.11	0.004	0.00012	0.00005	<u>0.0052</u>	0.0018	0.00018	0.00005
Total Copper	0.007-0.0174	0.002-0.007	<u>0.0068</u>	0.004	**<u>0.47</u>**	**<u>0.058</u>**	**<u>0.028</u>**	<u>0.0031</u>
Total Iron	1	-	0.12	0.044	**8.8**	**2.5**	0.23	0.08
Dissolved Iron	0.35	-	0.015	0.015	0.049	0.023	0.015	0.015
Total Mercury	-	0.00001	0.0000025	0.0000025	<u>0.000012</u>	0.000010	0.0000025	0.0000025
Total Selenium	-	0.002	0.0010	0.0010	<u>0.0021</u>	0.0014	0.00028	0.00013
Total Zinc	0.03-0.09	0.01-0.06	0.0015	0.0015	0.023	0.013	<u>0.021</u>	0.0015

Concentrations shown are in units of milligrams per litre (mg/L). **Bold** = Concentration exceeds the maximum WQG. <u>Underlined</u> = concentration exceeds the 30-day average WQG

(a) BC Water Quality Guidelines (WQG; January 2017) are shown for maximum (acute) and 30-day average (chronic or long-term) guidelines. The guidelines presented represent the ranges applicable to all results shown. The guidelines for total copper and zinc are hardness dependent. Exceedances are based on the average hardness (expressed as mg/L $CaCO_3$) for each water body, for samples collected between 1 May and 30 November 2015 (average hardness for Polley Lake = 130 mg/L; Hazeltine = 164 mg/L; Quesnel Lake = 54 mg/L).

(b) Results are shown for sampling dates between 1 May and 30 November 2015 (number of samples used in calculations: Polley Lake n = 81, Hazeltine n = 51, Quesnel Lake n = 213, with the exception of mercury [n = 12, 12, and 37, respectively]).

[9] Background conditions, defined as prior to the TSF embankment breach, were provided in Section 3.1 of Appendix F of MPMC 2015. Background data were available for Polley Lake, the Hazeltine Channel, and Quesnel River (limited metal data for Quesnel River). Background data for Quesnel Lake data were limited to nutrient-related monitoring surveys primarily focused around Horsefly Bay. Insufficient data were available to characterize background conditions for Quesnel Lake.

Many substances did not have a maximum concentration that exceeded the ambient WQG, and therefore, were not summarized in Table 6. Substances that had a maximum concentration that exceeded the chronic ambient WQG in any of the three waterbodies between May and November 2015 are summarized, of those, only three substances (copper, iron, and chromium) also had a 95th percentile that exceeded the chronic ambient WQG. Copper was retained as a COPC because of its known relationship to tailings, but iron and chromium were reviewed in greater detail.

- The maximum and 95th percentile concentrations for total iron exceeded the maximum WQG in samples collected from the Hazeltine Channel. Golder reviewed the data and found that total iron exceeded the maximum WQG on only four occasions between May 1 and November 30, and in each event, total suspended solids were also detected (36 to 557 mg/L). Concentrations of dissolved iron on all four occasions were less than the detection limit (<0.030 mg/L). As a result, Golder concludes that the four exceedances with respect to total iron are more reflective of turbidity events and do not indicate that iron is a substance of toxicological interest.
- The maximum and 95th percentile concentrations of total chromium exceeded the chronic 30-day WQG in samples collected from the Hazeltine Channel. This 95th percentile was calculated based on the entire data set of samples collected between May 1 and November 30. This provides a conservative basis for screening out COPCs, but ultimately, the decision about whether a chronic 30-day WQG was exceeded is based on the average concentration over a 30-day period. Golder has reviewed the data in greater depth and concludes that chromium is not a COPC as the rolling 30-day average was consistently less than the chronic WQG.

The DSI concluded that in the first few months after the TSF embankment breach, an initial increase in turbidity and concentrations of total metals in surface water were observed which extended to the mouth of Quesnel River. The 95th percentile concentrations of suspended particulates (i.e., total suspended solids or turbidity) in Hazeltine Channel also exceeded the ambient maximum and 30-day WQGs during the time frame (1 May to 30 November) considered for the purposes of selecting COPCs; whereas concentrations were below WQGs in Polley and Quesnel Lakes. Copper concentrations exceeded the ambient water quality guideline in Hazeltine and Edney Creeks. Copper concentrations in Polley and Quesnel Lakes decreased over time and did not exceed the short-term exposure (maximum) ambient WQGs by the end of 2015, but did exceed the long-term exposure 30-day chronic[10] ambient WQGs in some samples.

Copper was therefore retained as a COPC in the aquatic risk assessment. The trends in total suspended solids and turbidity over time were discussed in the DSI (Appendix A) and showed that suspended particulates have declined over time to concentrations that are well below the applicable guideline values. As such, turbidity does not need to be evaluated as a separate stressor in the risk assessment, but was retained as a potential confounding factor in how some lines of evidence (e.g., toxicity testing) were evaluated. Suspended particulates would also influence the total copper concentrations, as increased total copper was correlated with turbidity, thereby reflecting suspended particulate. Therefore, both total and dissolved copper concentrations were considered in the exposure assessment.

[10] Applicable to the average of five samples collected weekly over 30 days.

2.4.2.3 Review of the DSI Sediment Chemistry With Respect to Aquatic COPCs

Sediment chemistry data considered for the purposes of identifying COPCs were available from the following technical appendices prepared by Minnow Environmental Inc. (Minnow), and ultimately considered in the DSI (Golder 2016b, Appendix A).

- The initial sediment quality impact characterization was undertaken as part of the PEEIAR (MPMC 2015). This data set consisted of results from 75 stations sampled between August and October of 2014 (i.e., within three months of the TSF embankment breach) from impacted or potentially impacted areas of the Hazeltine Channel, Polley Lake, and Quesnel Lake (littoral and profundal zones). A further 31 stations were sampled in reference locations in Bootjack Lake or Quesnel Lake[11]. The general approach for this sampling program was to collect up to five discrete grab samples to create a composite sample for each station[12]. Stations were clustered in representative areas to facilitate statistical comparisons among locations.
- The supplemental sediment quality data report was prepared as part of the DSI (Appendix A-7.1). This data set consisted of results from 25 stations sampled from the Hazeltine Channel, Polley Lake, and Quesnel Lake, as well as from 13 reference stations in Bootjack Lake and Quesnel Lake[13]. Samples were collected in August 2015 following the same approach as the original data set described above; the objective was to duplicate the sampling at targeted locations to examine potential changes in sediment chemistry between 2014 and 2015.

The sediment chemistry data were used to select COPCs as part of the DSI. The 90th percentile concentrations were determined for each impacted area as well as the reference (Appendix A-7.2). A substance was retained as a COPC if it exceeded the screening value (see below) and the reference concentration.

Selection of the Screening Value

The primary screening tool for selecting COPCs was the CSR Schedule 9 sediment standards for sensitive sites ($SedQC_{SS}$). These standards apply to the top 1 m of sediment at sites that contain aquatic habitats important to fish spawning or serve as important rearing habitat for fish, or aquatic habitats used by species that are endangered, threatened or warrant special concern[14]. Golder has opted to conservatively apply the sensitive standards to all sediment irrespective of depth or whether or not the sensitive uses noted above are active. The provincial working sediment quality guidelines were considered on a case-by-case basis in the absence of a $SedQC_{SS}$ or CCME probable effects level (PEL) value.

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[11] See Table 4.2 of MPMC 2015 Appendix E for a list of specific sample locations where sediment chemistry samples were collected as part of the PEEIAR.

[12] See Section 4.1.of MPMC 2015 Appendix E for a discussion of sampling and analytical methods.

[13] See Figure 1 and Table A.1 of Appendix A-7.1 for a list of specific sample locations

[14] The application of sensitive standards to the ecologically active zone (i.e., the top 1 m of stable sediment) is described in Technical Guidance 15 to the Contaminated Sites Regulation. The definition of sensitive habitat is described in Procedure 8 of the Contaminated Sites Regulation.

The rationale for this screening value hierarchy is as follows:

- The $SedQC_{SS}$ is the primary screening value because it was developed from a statistical analysis of a data base of co-located chemistry and toxicity test data from field-collected sediments. The sensitive standards correspond to the chemical concentration that results in a 20% probability of observing a statistically significant difference between an individual sample and its corresponding negative control—they are intended to provide a conservative basis below which there is a low probability of observing an adverse effect in chronic toxicity tests with a variety of different test organisms[15]. These values are also promulgated in the provincial regulation that is specifically intended to be applied to contaminated sites.

- Screening limits from other jurisdictions compiled in BC MoE working sediment quality guidelines (BC upper working sediment quality guidelines) were considered in the absence of a $SedQC_{SS}$ value. Often, the upper working sediment quality guidelines adopts the CCME PEL value which is also based on a statistical analysis of a co-located database. However, BC working sediment quality guidelines have not yet been fully assessed or formally endorsed by the BC MoE, are not necessarily based on cause-effect studies, and should be applied with caution (BC MoE 2015). A minor exceedance of the BC working sediment quality guidelines is not considered likely to present a significant toxicological hazard to aquatic life. BC working sediment quality guidelines used to evaluate iron, manganese, and nickel were based on Jaagumagi (1993) which uses a different derivation approach.

  - The database consists of co-located chemistry and benthic community data from the Great Lakes following Persaud et al (1993).

  - Individual chemistry samples in the data set were rank-ordered for each substance of interest for each benthic species. For example, all samples where the amphipod *Hyalella azteca* was found to be present were sorted in terms of increasing iron concentration. The 90^{th} percentile of that distribution was used to set the maximum tolerance of that individual species to the given substance of interest. This process was repeated for nearly 100 different species that were identified in the cumulative benthic community data set.

  - The distribution of those threshold concentrations was then rank-ordered a second time and the numerical guideline set to the 5^{th} percentile of the ranked threshold values. For example, the iron guideline was set to the value that was lower than the maximum tolerance value for 95% of all the species considered.

  - This approach leads to a highly conservative screening value because it does not consider how physical factors (depth; habitat disturbance; changes in grain size or organic carbon) could influence the presence or absence of certain species in the data set.

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[15] See Chapter 5 of MESL (2003) "Development and applications of sediment quality criteria for managing contaminated sediment in British Columbia. Prepared for the Ministry of Environment by MacDonald Environmental Services Limited. Available online: http://www2.gov.bc.ca/assets/gov/environment/air-land-water/site-remediation/docs/policies-and-standards/develop_applicat_sqc_rep_nov19_wma.pdf

Identification of Sediment Contaminants of Potential Concern (COPCs)

This section provides a summary of the substances of interest evaluated during this process along with a rationale for their inclusion or exclusion as COPCs. Sediment quality was not assessed in Quesnel River because this habitat is erosional and no areas of sediment deposition were identified during sampling.

- Selected – Copper and arsenic are selected as COPCs. The maximum and 90th percentile concentrations of these substances in Polley Lake, the Hazeltine Channel, and Quesnel Lake exceeded the $SedQC_{SS}$.
- Not retained – No exceedances of screening value. The maximum concentrations of cadmium, lead, and mercury did not exceed the $SedQC_{SS}$. The 90th percentile concentrations for chromium, manganese, silver, and zinc also did not exceed $SedQC_{SS}$.
- Not retained – No exceedance of background concentrations. The 90th percentile concentration of nickel in Polley and Quesnel Lakes exceeded the BC working sediment quality guidelines of 16 mg/kg, but was lower than background concentrations.
- Not retained - Weak toxicological basis. Iron had a 90th percentile concentration in both lakes that exceeded the BC WQSG of 21,200 mg/kg. The 90th percentile concentration also exceeded the background concentration for Polley Lake, but not in Quesnel Lake, indicating that exceedances of the screening value can occur from natural sources. Iron was also considered unlikely to be of toxicological concern.

2.4.2.4 Review of the Tissue Chemistry With Respect to Biomagnifying COPCs

Based on screening to guidelines for sediment and water, selenium and mercury were not identified as COPCs related to the TSF embankment breach, but these substances are discussed below for the purpose of considering metals/metalloids that have potential to biomagnify and impact health of aquatic organisms. To evaluate these substances, evaluations of tissue concentrations in resident aquatic organisms were conducted, which provide an improved basis for screening of concentration data relative to use of environmental quality guidelines in sediment and water.

Zooplankton Tissue Chemistry

To characterize aquatic biota inhabiting the water column of the lakes, ongoing sampling of zooplankton communities was conducted during the 2015 open-water period (open-water habitats of Quesnel and Polley Lakes from May to September 2015). Samples were collected from three stations in Quesnel Lake (QUL): one exposed station called Hazeltine (QUL-Zoo-1; in the West Arm west of Cariboo Island) and two reference stations, Horsefly (QUL-Zoo-7; near the Horsefly River mouth) and Junction (QUL-Zoo-8; in the main basin where the east and north arms meet). Samples were also collected from two stations in Polley Lake: P1 and P2 (Appendix M-1).

Quesnel Lake

- Mercury—Concentrations of mercury in zooplankton tissue were variable at all three stations, with no consistent spatial or temporal trends. In several samples, mercury concentrations were below detection limits because of elevated detection limits from high moisture content in the samples.

- Selenium—Throughout the 2015 sampling period, concentrations of selenium in zooplankton tissue at Hazeltine were observed to be higher than tissue concentrations at Horsefly and Junction stations. Selenium tissue concentrations were generally below the BC interim dietary guideline for tissue consumption by fish (BC MoE 2014). In July and September 2015 zooplankton tissue concentrations of selenium measured at Hazeltine and Junction were above the dietary guideline. The highest concentration was observed at Hazeltine in September 2015 with a value of 9.53 mg/kg dry weight (dw).

Polley Lake

- Mercury—Concentrations of mercury in zooplankton tissue were generally similar between sampling stations (P1 and P2) in Polley Lake in 2015 with the exception of the September sampling event when P1 was higher.
- Selenium—Concentrations of selenium in zooplankton tissue were generally similar between sampling stations (P1 and P2) in Polley Lake in 2015 with the exception of the September sampling event when P1 was higher. Selenium tissue concentrations were at or above the BC interim dietary guideline for tissue consumption by fish (BC MoE 2014) at both stations during the sampling events in Polley Lake in 2015.

Due to laboratory processing errors and lack of replication in zooplankton tissue chemistry samples, the ability to characterize natural variability and identify discernible trends in zooplankton tissue chemistry is limited. Field replication and evaluation of sample collection protocols would be helpful to characterize the normal range of variability in this biological matrix and evaluate the usefulness as a long-term monitoring tool.

Fish Tissue Chemistry

Fish tissue chemistry data collected subsequent to the TSF embankment breach are reported in Appendix D. The data were divided into fish collected from exposed and reference areas, and subdivided by species, tissue type, and year to facilitate comparisons. For screening tissue data, the 2015 and 2016 data provided by Minnow are emphasized; these data were collected during recent sampling programs collected based on *a priori* sampling plans that are more balanced in terms of species collected, areas sampled, and number of samples.

Selenium—The BC MoE has the following selenium tissue guidelines for the protection of aquatic life: 11 mg/kg dw in egg/ovary tissues and 4 mg/kg dw in muscle and whole-body samples (BC MoE 2016). From a toxicological perspective, the preferred tissue for evaluating the potential for adverse effects to fish populations is eggs and ripe ovary tissues (McDonald and Chapman 2007; Janz 2012). BC MoE (2014a) derived the guideline of 11 mg/kg dw by applying a 2-fold safety factor to a toxicological effects threshold of approximately 22 mg/kg dw observed for cold-water salmonids such as rainbow trout. This guideline is intended to provide a protective screening value in the absence of a detailed site-specific evaluation.

- In Polley Lake, mean concentrations of selenium in muscle of rainbow trout and longnose sucker, and whole-body of redside shiner exceeded the tissue residue guidelines of 4 mg/kg dw by less than 2-times (Figure 11 of Appendix D-2).

- Rainbow trout collected from Frypan Creek in 2015 showed elevated concentrations of selenium in ovary that appeared anomalous relative to other species, locations, and years. These elevated concentrations were not observed in rainbow trout ovary samples collected in subsequent years from within Polley Lake. In 2016, selenium concentrations in rainbow trout ovary were higher than pre-breach data (2009-2012), whereas concentrations in longnose sucker ovary did not differ from pre-breach. Mean concentrations of selenium in ovaries of both species were 10 mg/kg dw in 2016, below the tissue residue guidelines of 11 mg/kg dw (Figure 11 of Appendix D-2).
- Muscle or whole-body selenium concentrations in fish collected from Quesnel Lake in 2015 and 2016 were below the tissue residue guidelines of 4 mg/kg dw in both reference and exposed areas (Figure 15 of Appendix D-2). All ovary samples were below the tissue residue guidelines of 11 mg/kg dw, consistent with previous observations (Appendix D-1).

Mercury—Mercury is primarily accumulated in muscle tissue of fish. There are no tissue residue guidelines from BC MoE for mercury; therefore, muscle and whole-body concentrations were compared to toxicity reference values (TRVs) from the literature. A wet weight benchmark of 0.5 mg/kg was derived from these studies that is applicable to both whole body and fillet (muscle) tissues, expressed on a wet weight basis.

- Beckvar et al. (2005)—The authors applied a systematic method for deriving protective (i.e., unlikely to have adverse effects) tissue residue-effect concentrations in fish using decision rules that were formulated to provide guidance on selecting studies and obtaining data in a consistent manner. Paired no-effect and low-effect whole-body residue concentrations in fish were identified for mercury and four analytical approaches of simple ranking, empirical percentile, tissue threshold-effect level, and cumulative distribution function were explored. A whole-body mercury tissue threshold-effect level of 0.2 mg/kg wet weight (ww) was derived, based largely on sublethal endpoints (growth, reproduction, development, behaviour), and was determined to be protective of juvenile and adult fish. The difference between the tissue threshold-effect level of 0.2 mg/kg ww and the selected benchmark of 0.5 mg/kg ww is partly attributable to the inclusion by Beckvar et al. (2005) of behavioural test endpoints (such as foraging behaviour and predator avoidance) and partly to a recognized "consistent bias toward protective tissue concentrations."
- Wiener and Spry (1996)—The authors summarize the information on mercury toxicity to fish available up to the mid-1990s. The review indicated that concentrations of total mercury associated with lethal and sublethal effects in freshwater fish were expected in the concentration range of 5 to 10 mg/kg ww in whole fish, with signs of overt toxicity not expected below this range. Although it was suggested that sensitive species may require a threshold in the low mg/kg range for protection against ecological impairment, there was no indication of adverse responses below 1.0 mg/kg ww.

- Sandheinrich and Wiener (2011)—The authors summarize a range of fish toxicity studies including evidence for mercury responses associated with biochemistry, gene expression, behaviour, reproduction, histology, and growth endpoints. This compilation serves as an update to the Wiener and Spry (1996) publication, and emphasizes that recent research has led to a reassessment of the concentration threshold previously thought to be protective. The authors conclude, based on an examination of a wide array of laboratory experiments and field studies, that freshwater fish are adversely affected at tissue concentrations of methylmercury below 1.0 mg/kg ww. They conclude that changes in biochemical processes, damage to cells and tissues, and reduced reproduction can occur at methylmercury concentrations of approximately 0.3 to 0.7 mg/kg ww in the whole body, and about 0.5 to 1.2 mg/kg ww in axial muscle tissues. The midpoint of the whole body threshold range proposed by Sandheinrich and Wiener (2011) is equal to the benchmark proposed for this assessment of 0.5 mg/kg ww. Furthermore, the lower end of the 0.3 to 0.7 mg/kg ww range appears to be associated with endpoints that have more uncertain ecological relevance. For example, the data presented in Sandheinrich and Wiener (2011) suggests that behavioural responses occur at concentrations slightly lower than reproduction, growth, and developmental responses. Furthermore, biochemistry and gene expression responses can occur at low concentrations, but these responses are indicators of environmental exposure that do not necessarily translate into ecologically significant effects.

On this basis, the proposed 0.5 mg/kg ww threshold for whole-body fish tissue concentrations was retained as a protective chronic effects benchmark for the updated assessment. The upper-bound (maximum and 95th percentile) concentrations from the most recent fish tissue data report (Appendix D-2) were compared to this screening value, yielding the following conclusions:

- For whole-body measurements of peamouth chub, redside shiner, and northern pikeminnow (*n*=24), all fish collected in Quesnel Lake near Hazeltine Channel mouth exhibited concentrations below 0.5 mg/kg ww (the maximum individual concentration was 0.61 mg/kg dw, or approximately 0.12 mg/kg ww).
- For muscle tissue measurements of rainbow trout and longnose sucker (*n*=33) collected in Polley Lake, the measured concentrations were below 0.5 mg/kg ww (the maximum individual concentration was 0.13 mg/kg ww).
- For most species sampled in Quesnel Lake (near Hazeltine), including burbot (*n*=4), largescale sucker (*n*=16), and peamouth chub (*n*=8), maximum concentrations of mercury in muscle tissue were below the 0.5 mg/kg ww benchmark (the maximum individual concentration was 0.48 mg/kg ww for a burbot muscle tissue sample)
- For lake trout sampled in Quesnel Lake (near Hazeltine), the maximum concentration (0.53 mg/kg ww) and 95th percentile (0.51 mg/kg ww) marginally exceeded the 0.5 mg/kg ww benchmark, based on analysis of 10 fish. Sandheinrich and Wiener (2011) indicate that mercury concentrations of about 0.5 to 1.2 mg/kg ww in axial muscle tissues can exhibit biological responses; however, at the lower end of this range the responses tend to be biochemical and cellular responses, rather than the growth, reproduction, and development endpoints used more commonly in risk assessment.

Implications for Risk Assessment

The analysis of tissue concentrations of mercury in resident biota indicate that mercury is not a constituent of concern. Neither the comparisons in bioaccumulation among stations, nor the screening to effects-based tissue screening benchmarks, indicate that the tissue burdens pose a significant environmental risk.

For selenium, there are indications of tissue exposure that are slightly above interim benchmarks for protection of aquatic life. Selenium has previously been identified as a constituent of interest in the regional environment even prior to the TSF embankment breach, and there was already ongoing monitoring of selenium in fish tissue (Minnow 2014). The approach for deriving site-specific science-based benchmarks for quantitative risk assessment of selenium is an evolving topic. The screening above is not considered a comprehensive analysis that is required for risk assessment of selenium per McDonald and Chapman (2007).

In recognition of the minor exceedances of screening benchmarks for tissue bioaccumulation, and the inconclusive results from the spatial evaluation of selenium concentrations in aquatic organism tissue, this substance cannot be definitively excluded from further assessment and is part of ongoing monitoring. The question of selenium management is one that the Mine was working on in the context of mine operations. However, given the scope of this risk assessment (i.e., focused on effects associated with the TSF embankment breach), selenium will not be considered further in this document. This decision is procedural and is not intended to be dismissive of the importance of the pre-breach and current findings.

2.4.2.5 Final Selection of COPCs for the Aquatic Risk Assessment

The screening process identified the following substances as COPCs for the aquatic ecological risk assessment:

- Copper: in surface water and sediment
- Arsenic: in sediment

Identification of copper and arsenic as the priority substances of interest for evaluation in the risk assessment does not mean that other metals or metalloids were not evaluated. Many of the lines of evidence involve measurement of biological or ecological effects as a result of exposure to the mixture of metals and metalloids that are present in tailings. Focusing on priority substances such as copper and arsenic is appropriate and consistent with risk assessment practice, particularly when evaluating substances that have toxicologically-based screening values that have been adopted as part of provincial regulation regarding the management of contaminated sites.

2.4.3 Aquatic Receptors of Potential Concern

This section highlights receptors of potential concern for the aquatic risk assessment. There were no rare, threatened, or endangered species identified in the study area. However, Interior Coho salmon (*Oncorhynchus kisutch*), are a locally present endangered regional race of this salmon species (DFO 2014, McRae et al. 2012). There are no reasons to believe that this race of Coho salmon are more or less sensitive to the COPC than other species. The risk assessment focuses on representative species selected from different groups of organisms (e.g., *Ceriodaphnia dubia* is a representative species for the broader zooplankton ecosystem component) based on consideration of the site's ecology. Some risk assessment tools focus on the entire ecosystem component rather than an individual surrogate species. The potential study area for the aquatic risk assessment included a variety of aquatic habitat types:

- **Creeks**—Examples of this habitat type include first and second order creeks such as the Hazeltine Channel and Edney Creek. The baseline characteristics of the Hazeltine Channel are representative of this habitat type. The Hazeltine Channel consisted of riffle-runs and shallow pools in a well-defined channel dominated by gravels and cobbles with a typical gradient of <2%. Steeper sections consisted of step-pools dominated by larger substrates types (Minnow 2014). A typical mean discharge rate was on the order of 0.19 cubic metres per second (m^3/s) (Minnow 2014). Benthic invertebrates in the Hazeltine Channel had a high proportion of sensitive EPT[16] taxa (mayflies, stoneflies, and caddisflies) (Minnow 2014). Fish in the upper portion of the Hazeltine Channel were dominated by rainbow trout (because of the presence of a barrier to fish migration), but also included a variety of other salmonids (various salmon species, kokanee, mountain whitefish) in the lower creek (MPMC 2015). Non-salmonid species in lower Hazeltine Channel included burbot, suckers, longnose dace, and peamouth chub (MPMC 2015). A similar assemblage of fish, as to lower Hazeltine Channel, was found in Edney Creek (MPMC 2015).
- **Lakes**—This habitat type includes Polley Lake and Quesnel Lake. These lakes have a littoral zone where aquatic plants are rooted and light can penetrate to the lakebed. Benthic invertebrates in Polley Lake had a high proportion of chironomids and oligochaetes, and the zooplankton community included rotifers, cladocerans, and copepods (Minnow 2014). Fish in Polley Lake included rainbow trout, longnose sucker, and red shiners. There are numerous fish species present in Quesnel Lake, including multiple salmon species, mountain whitefish, Dolly Varden, bull trout, and lake trout (MPMC 2015). Non-salmonid species include burbot, various species of suckers and dace, northern pikeminnow, sculpins, sturgeon, and lamprey (MPMC 2015). Early juvenile stages of salmon (*Oncorhynchus* spp.), burbot, and lake whitefish are benthivores (i.e., their food comes from benthic substrates). Although most fish species interact with the near-shore environment to some degree, Quesnel Lake also has substantial deep, open-water pelagic habitat that would not be directly connected to bottom sediment. Species that utilize this open-water habitat (e.g., juvenile sockeye salmon and kokanee) will feed on copepods and other zooplankton species or other fish, depending on their size.
- **River**—Quesnel River flows from Quesnel Lake to the Fraser River. The portion of Quesnel River within the study area consists of deep water glides (MPMC 2015) with a mean discharge rate on the order of 250 m^3/s (i.e., more than three orders of magnitude greater than the Hazeltine Channel). A variety of salmonid fish species use the Quesnel River, including rainbow trout, multiple salmon species, mountain whitefish, Dolly Varden, bull trout, and lake trout (MPMC 2015). Non-salmonid species include burbot, various species of suckers and dace, northern pikeminnow, sculpins, sturgeon, and lamprey.

[16] EPT = ephemeroptera – plecoptera – trichoptera.

The three habitat types share a common set of major receptor groups, even if the species within the receptor group varies. The major receptor groups for the aquatic risk assessment (along with representative taxonomic groups or species) are summarized in Table 7.

Table 7: Receptor Groups with Representative Taxonomic Group or Species for Aquatic Risk Assessment

Receptor Group	Trophic Level / Feeding Guild	Habitat Type and Representative Taxonomic Group or Species	
		Creeks or Rivers	Lake
Aquatic plants	Algae and periphyton	Community	Not applicable
	Vascular plants and mosses	Community	Not applicable
Benthos	Benthic invertebrates	Chironomids (*Chironomus* sp).	Chironomids (*Chironomus* sp).
	Epibenthic invertebrates	Stoneflies (Order Plecoptera)	Amphipods (*Hyalella azteca*)
Plankton	Zooplankton	Cladocerans (*Ceriodaphnia*)	Cladocerans (*Ceriodaphnia*)
Fish	Small bodied forage fish	Rainbow trout (*Oncorhynchus mykiss*)	Rainbow trout (*Oncorhynchus mykiss*)
	Large bodied piscivorous fish	Not applicable	Chinook salmon (*Oncorhynchus tshawytscha*)
Wildlife	Piscivorous birds and mammals	Great blue heron (*Ardea herodias herodias*) Osprey (*Pandion haliaetus*) River otter (*Lontra canadensis*)	

2.4.4 Identification of Exposure Pathways

There are several routes by which an aquatic organism can be exposed to a given contaminant, and the route of exposure often influences both the site of toxic action and toxicity mechanism for the organism (McGeer et al. 2003). Aquatic receptors described in Table 7 could be exposed to the COPCs via a combination of one or more of the following pathways:

- Direct exposure to sediment
- Ingestion of sediment
- Direct exposure to surface water
- Ingestion of food items

All four of these exposure pathways are operable, but the relative contribution of each pathway to risks for a given receptor is influenced by the environmental fate, transport, and bioavailability of each COPC under site-specific conditions. There is also the physical effect associated with the initial scouring and deposition of material during the breach which is an additional stressor that needs to be considered in the interpretation of each individual line of evidence.

These four main exposure pathways are described below for each of the major receptor groups in the context of how the environmental fate, transport, and bioavailability of copper (and to a lesser degree arsenic) under site-specific conditions will likely influence the relative contribution of each operable pathway to risks for a given receptor. This discussion is meant to highlight potential factors that may influence the relative magnitude of the different operable exposure pathways. An appreciation of these important factors helps improve the realism of the problem formulation, and by extension, improves the linkage between the selected lines of evidence and the overall assessment endpoints.

Uptake by Sediment-Associated Receptors

Sediments are a depositional “sink” because particulates (organic carbon and inorganic minerals, including tailings) tend to settle to the bottom of waterbodies. Although some redistribution happens over time through resuspension and transport in accordance with the underlying geomorphology of the site, the majority of the mass remains in the sediment. Sediment transport in the littoral areas along the shoreline is influenced by wind-driven waves, but is reduced in importance in deep water areas where sediment would have higher stability and tendency to form depositional layers. The bioavailability of metals in sediment depends on its tendency to desorb from tailings into sediment porewater. There are multiple geochemical mechanisms involved including complexation with dissolved organic carbon, precipitation as metal-sulphide complexes, and adsorption and binding to colloids and particulates (Chapman et al. 2003). Metal bioavailability under field conditions is often less than under laboratory conditions (Janssen et al. 2003), and from a risk assessment perspective, the exposure parameter of greater interest is the bioavailable fraction of the total metal concentrations. Porewater concentrations (and selective leach analyses) often provide a more realistic measure of the bioavailable fraction relative to bulk sediment concentrations. The substrate types and sediment transport dynamics also influence which taxonomic groups of benthic organisms will be dominant in an area. Benthic organisms (infauna) tend to burrow into the sediment and are exposed to metal via direct contact (with sediment porewater and sediment) as well as sediment ingestion, although they can also circulate overlying water in their burrows. Epibenthic organisms occupy the sediment-water interface and are exposed via sediment ingestion, respiration of water, and to a lesser degree, to porewater fluxing into the water column.

Uptake by Water Column Dwelling Organisms

The environmental fate of metals in the water column is affected by a variety of environmental and physiochemical influences, including the physical transformation, chemical fate properties (e.g., adsorption to smaller particulates, sulphide and dissolved organic matter interactions), and hydrological processes. Primary exposure routes include:

- Dissolved phase exposure (uptake through respiratory surfaces or the skin)
- Particulate exposure (metals bound to particulates in suspended solids)
- Dietary uptake (ingestion of pelagic prey, such as zooplankton feeding on phytoplankton, or juvenile fish feeding on plankton)

The degree of uptake of metals by aquatic biota from these pathways via the dissolved phase is strongly influenced by bioavailability and/or transfer processes such as ligand binding and receptor site competitive interactions (McGeer et al. 2003). These interactions for copper are relatively well-understood and models to integrate the known mechanisms of geochemical speciation, metal–organic binding, and the physiological mechanisms of toxicity have been developed (i.e., biotic ligand modelling; Smith et al. 2015). The toxicity of copper and other metals is dependent on a suite of factors, including the chemical form of the metal, other water quality characteristics that affect bioavailability (e.g., dissolved organic carbon, pH, and temperature), the duration and magnitude of metal exposure, and the relative sensitivities of aquatic species.

Internal Toxicokinetics of Metals

Metals can bioaccumulate in the tissues of aquatic organisms once they have been absorbed or ingested through one or more of the exposure pathways described above. There is no evidence that copper biomagnifies in aquatic systems. Site-specific accumulation factors suggest that concentrations are the highest at the base of the food chain. Once ingested or absorbed, the partitioning of the metal into specific tissues within the organism depends on a number of factors, including internal sequestration, storage, elimination kinetics, and nutrient essentiality. The uptake of metals differs from organic COPCs in that aquatic biota are able to regulate internal concentrations of metals (McGeer et al. 2003), and the uptake of metals ions tend to be rate limited (i.e., rates of metal uptake decline as metal exposures increase). This results in an inverse relationship between bioconcentration factor and exposure concentration for metals in aquatic biota (McGeer et al. 2003). Adams (2011) highlights this inverse relationship for several metals, including copper, in a range of invertebrates such as amphipods (e.g., *Hyalella azteca*), polychaetes (*Phyllodoce maculata*, *Eudistylia vancouveri*) and mollusks. In addition to processes that regulate the uptake of metals, there are processes that sequester, eliminate, or detoxify metals once they are accumulated. Freshwater fish, such as rainbow trout, actively regulate copper via metal sequestration into the liver and elimination via the bile, a process that involves copper-specific transport mechanisms (McGeer et al. 2003). Detoxification of copper through binding to metallothionein-like proteins has been documented as of significance in both marine and freshwater organisms. Detoxification and storage of copper in granules has also been observed (Smith et al. 2015).

2.4.5 Physical Influence of Tailings and Debris

The physical effect of the tailings and debris release on aquatic organisms and their habitat is an important consideration in the current risk assessment. Physical alterations such as pronounced burial or scour (similar to what can occur during a landslide) can cause short-term elimination of invertebrates and/or their habitat as well as affect fish access to habitats that may be important for various life stages. As noted above, suspended particulates were identified as a stressor of concern that can exert negative effects through one or more of the following mechanisms:

- Excess suspended sediments in the water column impair respiration, feeding, and visual foraging of aquatic biota (Waters 1995; Wood and Armitage 1997).
- The deposition of excess fine sediment decreases substrate particle size, resulting in a change from a heterogeneous substrate with complex habitats to a more homogenous sand/silt substrate. Fine-grained sediment can clog interstitial spaces (Waters 1995, Zanetell and Peckarsky 1996), reduce substrate stability (Cobb et al. 1992, Jowett 2003), and increase substrate embeddedness (Sylte and Fischenich 2002).

Suspended particulate concentrations have declined to less than the applicable WQGs and therefore, they are unlikely to be a stressor of potential concern under current conditions. However, the confounding effect of suspended solids is a relevant consideration for some of the effect-based data compiled for this risk assessment.

2.4.6 Aquatic Conceptual Site Model

The conceptual site model (Figure 5) integrates the source of COPCs, operable exposure pathways, and receptors of potential concern into a single figure. The purpose of the conceptual site model is to distill the available information into a more simplified depiction of the key risk pathways to help convey the associated hypotheses concerning the nature of ecological risks at the site.

2.4.7 Application of the Weight-of-Evidence Framework

The weight-of-evidence framework was introduced in Section 1.2.2. The weight-of-evidence framework helps to provide clarity about what types of information are available for the risk assessment, and how such information will be evaluated. The framework also helps to determine how much "weight" or strength a particular line of evidence will get in the overall risk conclusion. The aquatic risk assessment considered the following assessment endpoints and major lines of evidence:

- Assessment Endpoint 1: Protect aquatic receptors from potentially hazardous metals exposures via direct contact with water and sediment. Lines of evidence related to this assessment endpoint focus on comparing the total concentrations of COPCs to risk-based numerical guidelines, standards, or screening values, as well as surrogate measurements of how much of the total concentration is likely to be mobile or bioavailable (e.g., leaching potential from tailings; sequential extractions; bioavailability models such as acid volatile sulfide - simultaneously extracted metals [AVS-SEM]).
- Assessment Endpoint 2: Protect benthic organisms from impairment of functional health attributes (e.g., survival, growth, reproduction, or normal development) as a result of direct contact with sediment. Lines of evidence related to this assessment endpoint include measurements of effects in the field or in the laboratory, or provide a direct evaluation of bioavailability (e.g., diversity and abundance of resident benthic organisms, toxicological effects in standardized laboratory toxicity tests, tissue bioaccumulation comparisons between exposed and reference areas).
- Assessment Endpoint 3: Protect plankton from reductions in survival, growth, and reproduction as a result of direct contact with water. This assessment endpoint is similar to the one described above for benthic organisms and incorporates lines of evidence such as direct measurement of zooplankton diversity and abundance and phytoplankton biomass in the field, as well as measurement of adverse effects in the laboratory using standardized toxicity tests.
- Assessment Endpoint 4: Protect fish from reductions in impairment of functional health attributes (e.g., survival, growth, reproduction, or normal development) as a result of direct contact with abiotic media or ingestion of prey items. This assessment endpoint includes several lines of evidence intended to determine if adverse effects are occurring to individual fish or to fish populations. Lines of evidence related to this assessment endpoint include consideration of metals concentrations in prey items, concentrations bioaccumulated into fish tissues (i.e., the sum of exposure from prey items and water), standardized toxicity tests, direct measures of fish disease (histopathology), and direct measures of individual and population condition (e.g., condition factor and demographics).

The specific lines of evidence for each of these major assessment endpoints are described in further detail in Section 4.0.

3.0 TERRESTRIAL ECOLOGICAL RISK ASSESSMENT

3.1 Introduction

3.1.1 Review of the Problem Formulation

Section 2.3 highlighted that the purpose of the risk assessment was to determine if the contaminant stressor (i.e., residual concentrations of metals in tailings) represent an unacceptable risk that requires risk management over and above the rehabilitation and monitored recovery program that is already underway to address the physical stressors inherent from the scouring and deposition of tailings. Impacts to the terrestrial community as a result of those physical stressors were substantive. The spatial extent of these impacts was the footprint where tailings were deposited and/or land was scoured. Further specific lines of evidence to quantify this impact are not necessary for the purpose of this ecological risk assessment.

The management goal is to achieve the continued presence (or successful reintroduction) of a biologically diverse, functional, self-sustaining and inter-dependent ecosystem. The available site information was reviewed to identify COPCs, receptors of potential concern and viable exposure pathways. Based on this information, the following assessment endpoints and major lines of evidence were articulated:

- Assessment Endpoint 1: Protect microbes, soil invertebrates and plants from potential exposure from metals via direct contact with soil. This assessment endpoint focuses on understanding the potential hazards associated with total concentrations of the COPCs in the environment. It corresponds to the "exposure assessment" in the classic risk assessment paradigm. Lines of evidence related to this assessment endpoint focus on comparing the total concentrations of COPCs to risk-based numerical guidelines, standards or screening values, as well as surrogate measurements of how much of the total concentration is likely be mobile or bioavailable (e.g., leaching potential from tailings).
- Assessment Endpoint 2: Protect soil invertebrates and plants (collectively, the lower trophic levels) from reductions in survival, growth and reproduction as a result of direct contact with tailings. This assessment endpoint focuses on lines of evidence that measure effects in the field or in the laboratory, or provide a direct evaluation of bioavailability. Examples include a quantitative forest health assessment that looked at a number of metrics such as soil structure, average rooting depth and soil nutrient levels, documentation of vegetation species composition and percent cover, measurement of adverse effects in the laboratory using standardized toxicity tests; and comparison of the concentrations of metals in invertebrates and plants between exposed and reference areas. Many of these lines of evidence provide insight about the relative influence of contamination versus the potential long term influence of physical stressors.
- Assessment Endpoint 3: Protect terrestrial wildlife from reduction in survival, growth and reproduction as a result of exposure via ingestion of food items, incidental soil and drinking water. A variety of relevant receptor species from different feeding guilds (collectively, the higher trophic levels encompassing birds and mammals) was evaluated using a food chain model that compared the estimated daily ingested dose to the toxicologically safe concentration from the literature.

3.1.2 Content of this Chapter

This chapter provides a review of each individual line of evidence that was considered in the exposure and effects assessment stages of the terrestrial risk assessment. This review provides:

- A discussion of the study design and methodology used for data collection (including consideration of how the line of evidence met its underlying data quality objectives)
- A summary of the data obtained for the line of evidence during the course of the investigation
- A discussion of how the decision criterion for the line of evidence was developed and presentation of how the data compares to that decision criterion
- A discussion with respect to the uncertainty in the data
- A summary of the key findings that will be considered in the weight-of-evidence integration

The technical reports as prepared by the principal investigators for each study component are provided as appendices; these reports provide the detailed information on individual samples, specific analytical results, detailed methodologies, and quality assurance. Some technical reports were developed to address specific study questions at the time to address items that the public needed to know or to provide interim recommendations for preliminary risk management decisions, reflecting the overall adaptive nature of the approach used by the mine and by government (Nikl et al., 2016). A key objective of this report section is to aggregate the technical information from multiple rounds of testing, synthesize technical information from many different investigators, and align that information with provincial risk assessment guidance.

3.2 Exposure-Based Line of Evidence (LOE): Soil Chemistry

3.2.1 Bulk Soil Concentrations

Copper and vanadium were identified as COPCs in soil in the problem formulation (Section 2.3). The following sections provide further context regarding the distribution of these contaminants as part of the exposure assessment. This includes sampling conducted in December 2016 after the completion of rehabilitation activities in the Hazeltine corridor.

3.2.1.1 Description of Material

Post-breach conditions consisted of tailings mixed with varying amounts of native soils that were scoured from the Hazeltine channel. Tailings consisted of a mixture of two types identifiable based on visual properties (e.g., colour, texture), which were confirmed by geochemical properties (i.e., magnetism, field carbonate effervescence class, and visible mineralogy):

- Grey tailings with a fine silty texture which were predominant along the upper benches of the floodplain and in the forest halo zone.
- Black or orange magnetite sand which was predominant in low-lying areas near Hazeltine creek

The soil quality impact assessment (MPMC 2015) found that concentrations of copper and vanadium in tailings decreased slightly with distance from the tailings storage facility but were ultimately part of the same statistical population. There were no trends observed with regard to vertical stratification of concentrations within the deposited tailings[17]. The maximum thickness of the deposited tailings ranged from several centimeters of a predominantly ‘grey’ tailings in the halo areas to isolated areas with a thickness greater than 3 m of a mixture of ‘grey’ tailings and ‘magnetite’ tailings in portions of the Polley Flats. Tailings were not observed within the Hazeltine Canyon (except in thin, isolated pockets). The material deposited in lower Hazeltine Channel and at the mouth of Edney Creek tended to consist of a mixture of native till and tailings.

3.2.1.2 Updated Soil Exposure Sampling

Post-breach soil conditions were characterized in the DSI and DSI update reports (Golder 2016b; Appendix A). These data were used for the purposes of identifying the COPCs, but the Hazeltine corridor has been the subject of ongoing rehabilitation, and therefore, relying only on the older data would not necessarily provide the most realistic exposure information. Three different rehabilitation methods have been implemented:

- Transplanted—In these areas, tailings were removed down to the underlying native soils and soil transplants composed of topsoil and mineral soil stockpiled from the Perimeter Embankment and Main Embankment construction work were added. Transplanted soils zones are located mostly within the Polley Flats floodplain.
- Intermixed—In these areas, tailings were mechanically mixed with underlying native organic and mineral soils to increase atmospheric connectivity with the topsoil. Soils were overturned using an excavator bucket or dozer blades until the overlying tailings were visually mixed with underlying soils (e.g., the mixed mounds were visually checked for the darker organics soils). Intermixed soils zones are mostly located within the halo.
- Scarified—In these areas, slopes (which consisted of mostly glacial till and some tailings) were re-graded and re-sloped to improve stability and reduce erosion potential. Forest soil and organics from the adjacent forest at the crest of these slopes was pulled down and integrated into the mineral rich soil in areas slope stability was a concern. Soils were scarified to reduce compaction.

Rehabilitation has also included placement of coarse woody debris, removal of dead trees and vegetation, seeding of soils and mounding and ripping of tailings in halo areas with underlying forest floor soils (Appendix G-1). Table 8 provides an overview of the estimated area of rehabilitation treatments. The remaining area of the Hazeltine corridor that has not yet been rehabilitated is referred to as ‘untreated’.

[17] The initial soil quality assessment detected a potential pattern where higher concentrations in tailings tended to be located in the upper portion of the Hazeltine Channel, but this trend was not verified as part of the DSI.

Table 8: Overview of Soil Exposure Areas

Exposure Area	Description	Area (ha)	Number of Samples Collected for DSI	Number of Samples Collected December 2016
Untreated	No significant rehabilitation works	124	182	Not applicable
Background	Not impacted by deposition of tailings	NA	52 organic and 34 mineral	Not applicable
Transplanted	Rehabilitated by removing tailings and adding transplanted soil	9	Not applicable	30
Intermixed	Rehabilitated by mixing tailings with underlying native soils	49	Not applicable	37
Scarified	Rehabilitated by regrading, adding local forest soils and scarified to reduce compaction	33	Not applicable	23

DSI = Detailed Site Investigation; ha = hectares; NA = not available.

All samples were collected in accordance with standard practices, irrespective of whether the specific sample was collected for DSI delineation purposes, as part of other lines of evidence to support the risk assessment, or to provide additional context regarding soil concentrations in background or rehabilitated areas. A description of sampling and quality assurance and quality control methods are provided in the original reports, along with tabular summaries, sampling locations, original certificates of analyses, chain of custody forms, and supplemental information such as soil descriptions. All samples were analyzed for total metals using the strong-acid leachable metals protocol (BC MoE 2015b). Golder has reviewed these original reports to check that quality assurance/quality control practices were appropriate before the data were considered reliable for the purposes of the risk assessment.

3.2.1.3 Summary of Findings

Table 9 summarizes the concentration data for copper and vanadium for each exposure area. The DSI included calculation of a local background concentration using the general approach described in Technical Guidance 16. In brief:

- Rehabilitation resulted in reductions in copper and vanadium concentrations. This reduction is likely associated with the mixture of natural soils with lower concentrations, but ultimately, the bulk concentration in terms of mg metal per kg soil that is relevant to the exposure assessment is lower in the rehabilitated areas than in the original 100% tailings.
- Copper concentrations continue to exceed the calculated local and regional background concentrations as well as CSR standards for the protection of soil invertebrates and plants.
- Vanadium concentrations continue to exceed the calculated local and regional background concentrations as well as the generic CSR standard in the untreated samples, but do not exceed these screening values in the rehabilitated areas.

Table 9: Summary of Bulk Soil Chemistry of Contaminants of Potential Concern in Soil

Parameter		Soil Concentration	Applicable Numerical Standard[a]	Local Background (Organic)[b]	Local Background (Mineral)[b]	Regional Background[c]
Transplanted Soils Zone						
Copper	Maximum	532	150	79	39	65
	90th Percentile	302				
	95% UCLM	197				
Vanadium	Maximum	149	200	83	88	100
	90th Percentile	111				
	95% UCLM	95.7				
Intermixed Soils Zone						
Copper	Maximum	880	150	79	39	65
	90th Percentile	805				
	95% UCLM	538				
Vanadium	Maximum	216	200	83	88	100
	90th Percentile	170				
	95% UCLM	141				
Scarified Zone						
Copper	Maximum	715	150	79	39	65
	90th Percentile	368				
	95% UCLM	256				
Vanadium	Maximum	174	200	83	88	100
	90th Percentile	112				
	95% UCLM	99.6				
Tailings (Untreated Corridor)						
Copper	Maximum	1560	150	79	39	65
	90th Percentile	1000				
	95% UCLM	737				
Vanadium	Maximum	289	200	83	88	100
	90th Percentile	213				
	95% UCLM	177				

Concentrations are in milligrams/kilogram (mg/kg dw); UCLM = Upper Confidence Limit of the Mean;

(a) The applicable standard for copper is based on protection of soil invertebrates and plants for agricultural and parkland land uses (CSR Schedule 5). The applicable standard for vanadium is the generic standard for agricultural and parkland land uses (CSR Schedule 4).

(b) Local background is the 95th percentile concentration for organic and mineral samples collected from background. See the detailed site investigation for further information.

(c) Regional background values are from Protocol 4.

3.2.1.4 Comparison to Decision Criteria

Soil chemistry data are used in the risk assessment in a variety of ways:

- Soil chemistry summary statistics were compared to numerical standards as an exposure-based line of evidence (current section)
- Soil chemistry data were used as an input value in the food chain model in terms of ingested soil concentrations and to support the calculation of bioaccumulation factors to predict tissue concentrations in plants and soil invertebrates (Section 3.4.2). The data set was divided based on spatial considerations (e.g., each treatment type was considered separately, and the untreated area was divided into three units to improve the spatial resolution of the food chain model)
- Soil chemistry data were used to evaluate the relationships between concentrations and measured effects in toxicity tests (Section 3.3.3). This evaluation is based on the analytical chemistry data for the specific samples tested.

The calculated 90th percentile and 95% UCLM concentrations were compared to the pathway specific standard for the protection of soil invertebrates and plants (or generic, if no pathway specific standard was available) to provide an indication of the magnitude of hazards that may be present to soil invertebrates and plants. Further context was provided by comparing the 90th percentile and 95% UCLM concentrations to the local background concentration.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Soil chemistry	Protect microbes, soil invertebrates and plants from potentially hazardous exposure from metals via direct contact with soil	Compare calculated 90th percentile and 95% UCLM concentrations in soil to numerical standards	90th percentile and 95% UCLM does not exceed standard	Untreated portion of corridor	Does not meet decision criterion for Cu or V
				Treated portion of corridor	Does not meet decision criterion for Cu. Meets decision criterion for V

Risks are unlikely to be present if the reasonable worst-case estimates (such as the 90th percentile or 95% UCLM) do not exceed the standard or screening value. As a BC MoE policy, a site that does not contain exceedances of numerical standards would not be deemed to be a contaminated site under the *Environmental Management Act* and the CSR. Numerical standards are designed to provide a high degree of certainty that concentrations less than the standard will not pose a risk. A screening value may also have a similar degree of certainty depending on the specific circumstances of its derivation. Overall, numerical standards and appropriately conservative screening values help to focus the risk assessment onto specific COPCs, exposure pathways or areas that present the greatest potential for hazard. The presence of a hazard (i.e., a concentration that is greater than the standard or screening value) does not mean there is an actual risk—there are many modifying factors that influence bioavailability and toxicity (Lanno et al., 2004). A summary of the hazard quotients for copper and vanadium is provided in Table 10. In brief:

- The 90th percentile or 95% UCLM concentration for copper in tailings exceeded the numerical standard for the protection of soil invertebrates and plants by a factor of up to approximately 7. A hazard quotient of 7 cannot be ruled out as unlikely to be a hazard; further evaluation of the potential for adverse effects is warranted. The magnitude of the hazard for copper was higher when considered in the context of local or regional background values, highlighting that copper is associated with the tailings and that the deposition of tailings has resulted in an exposure concentration that is higher than what can be reasonably assumed to exist in the absence of the TSF embankment breach.

- The 90th percentile or 95% UCLM concentration for vanadium in tailings exceeded the generic numerical standard by a factor of up to approximately 1 (i.e., 0.9 or 1.1 depending on the summary statistic selected). A hazard quotient of near 1 suggests that hazards are unlikely to translate into unacceptable risks, however, magnitude of the hazard was higher when these concentrations were compared to the local or regional background values. This suggests that vanadium is associated with the tailings and that the deposition of tailings has resulted in an exposure concentration that is higher than what can be reasonably assumed to exist in the absence of the TSF embankment breach. Hazards for vanadium were also evaluated using a generic standard that is not necessarily based on a robust toxicological database. Golder has therefore opted to retain vanadium for further evaluation in the effects-based lines of evidence.

Table 10: Magnitude of Difference of Metals in Soil against CSR Standards and Background Concentrations

Parameter		Soil Concentration (mg/kg dw)	Magnitude of Difference of Soil Concentration to:			
			MCS AL/PL	Local Background (Organic)	Local Background (Mineral)	Regional Background
Transplanted Soils Zone						
Copper	Maximum	532	3.5	6.7	14	8.2
	90th Percentile	302	2.0	3.8	7.7	4.6
	95% UCLM	197	1.3	2.5	5.1	3.0
Vanadium	Maximum	149	0.75	1.8	1.7	1.5
	90th Percentile	111	0.56	1.3	1.3	1.1
	95% UCLM	95.7	0.48	1.2	1.1	0.96
Intermixed Soils Zone						
Copper	Maximum	880	5.9	11	23	14
	90th Percentile	805	5.4	10	21	12
	95 UCLM	538	3.6	6.8	14	8.3
Vanadium	Maximum	216	1.1	2.6	2.5	2.2
	90th Percentile	170	0.85	2.0	1.9	1.7
	95% UCLM	141	0.71	1.7	1.6	1.4

Parameter		Soil Concentration (mg/kg dw)	Magnitude of Difference of Soil Concentration to:			
			MCS AL/PL	Local Background (Organic)	Local Background (Mineral)	Regional Background
Transplanted Soils Zone						
Scarified Zone						
Copper	Maximum	715	4.8	9.1	18	11
	90th Percentile	368	2.5	4.7	9.4	5.7
	95% UCLM	256	1.7	3.2	6.6	3.9
Vanadium	Maximum	174	0.87	2.1	2.0	1.7
	90th Percentile	112	0.56	1.3	1.3	1.1
	95% UCLM	99.6	0.50	1.2	1.1	1.0
Tailings (Untreated Corridor)						
Copper	Maximum	1560	10	20	40	24
	90th Percentile	1000	6.7	13	26	15
	95% UCLM	737	4.9	9.3	19	11
Vanadium	Maximum	289	1.4	3.5	3.3	2.9
	90th Percentile	213	1.1	2.6	2.4	2.1
	95% UCLM	177	0.89	2.1	2.0	1.8

95 UCLM = 95% upper confidence limit of the mean.

3.2.2 Chemical Surrogates for Bioavailability in Soil

Total metal concentrations in soil are a poor predictor of toxic effects (e.g., Smolders et al. 2009, McLaughlin et al. 2000) because solubility, chemical speciation and bioavailability are not considered. There are a wide variety of chemical extraction methods described in the literature, and many studies have demonstrated statistically significant relationships between a given extraction method and other endpoints such as tissue concentrations, endpoints in laboratory-based toxicity tests, or field measurements. However, no single method adequately simulates how bioavailability varies with soil physiochemical properties for different receptors and metals. Chemical surrogate measurements provide information about a study-specific, operationally-defined “fraction of bioavailability” (Peijenberg et al. 2007). A number of different chemical surrogate measurements of bioavailability have been made during the course of the site investigation:

- Water soluble fractions from saturated paste extracts;
- Long-term leach tests using humidity cells and soil columns;
- Geochemical modelling
- Physiologically-based extraction tests

These surrogate measurements provide context about the degree that copper and vanadium in tailings would be bioavailable under field conditions. It was not considered necessary to conduct a battery of different extraction methods or a comprehensive review of the available literature to establish a rationale for the "best" extraction method under these circumstances. This type of refinement would be necessary only if the objective was to establish a statistically significant relationship between the extract concentration and other lines of evidence that involve direct measurement of effects.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Chemical Surrogates for Bioavailability	Protect microbes, soil invertebrates and plants from potentially hazardous exposure from metals via direct contact with soil	Qualitative evaluation of the overall leaching potential of tailings	Leaching potential is low	Site wide	Water soluble fraction of total concentration is low for both Cu and V
					Humidity cell and leach column test for Cu indicated low leachability. V not assessed.
					PBET found low bioaccessibilty for Cu, very low bioaccessibilty for V

3.2.2.1 Water Soluble Fraction Measured by Saturated Paste

A saturated paste extract (Carter and Gregoire 2006) was used to determine the water soluble fraction of metals in a soil sample. In brief, the method involves the addition of ultra-pure deionized water to an aliquot of approximately 300 grams of soil to form a paste (i.e., water is added only to the point to create the saturated paste). The paste is vacuum-filtered after four hours and the resulting extract is subjected to routine analysis of metals (inductively coupled plasma–optimal emission spectroscopy per EPA6010B). A subset of 35 soil samples collected as part of the post-rehabilitation soil investigation (see Section 3.2.1.2) were submitted for analysis. Samples were selected to represent a range of grain size and total organic carbon content in each of the three rehabilitation areas but were intentionally biased towards samples with higher metal content to provide a conservative assessment. Further details regarding sampling methods, station locations and the original certificates of analysis are provided in Appendix G-1. Copper and vanadium concentrations in the paste extracts were compared to provincial numerical standards.

The water soluble fraction of copper and vanadium were substantially lower than their respective total metal concentrations (Table 11) because total metal concentration are determined by a strong acid digestion which would tend to over-estimate the amount of metals that would be readily dissociable from tailings. Water soluble fractions of copper were typically less than 0.1% of the total concentrations. Water soluble concentrations of vanadium were less than the analytical detection limit of 0.3 mg/kg in all samples (i.e., were also less than 0.1% of the total concentration).

Table 11: Summary of Saturated Paste Extract Copper and Vanadium Concentrations

Parameter		Saturated Paste Extract Concentration	Total Metal Concentration	CSR Standard
Transplanted Soils Zone				
Copper	Maximum	0.13	532	150
	90th Percentile	<0.1	302	
	95% UCLM	-	197	
Vanadium	Maximum	<0.3	149	200
	90th Percentile	<0.3	111	
	95% UCLM	-	95.7	
Intermixed Soils Zone				
Copper	Maximum	1.2	880	150
	90th Percentile	0.50	805	
	95% UCLM	0.51	538	
Vanadium	Maximum	<0.3	216	200
	90th Percentile	<0.3	170	
	95% UCLM	-	141	
Scarified Zone				
Copper	Maximum	0.14	715	150
	90th Percentile	0.13	368	
	95% UCLM	-	256	
Vanadium	Maximum	<0.3	174	200
	90th Percentile	<0.3	112	
	95% UCLM	-	99.6	

Concentrations shown are in milligrams per kilogram (mg/kg).

95% UCLM = 95% upper confidence limit of the mean.

" –" = not calculated due to limited number of samples with detectable concentrations (i.e., only one or two samples).

Water soluble fractions have been used to evaluate the relationship between metal concentrations in soil and adverse effects measured in plants (Bagur-Gonzalez et al. 2011; Walker and Bernal 2004, Mench et al. 1994) and soil organisms (Boyd and Williams 2003; Lock et al. 2003). Other methods to simulate the bioavailable fraction of metals using a chemical surrogate are available (e.g., single extraction with diethylenetriaminepentaacetiac acid (DTPA); single extraction with $CaCl_2$) but the use of the water soluble fraction provides a reasonable approximation of the amount of copper that would likely dissociate from soil particles into soil porewater as a result of natural precipitation and thus be available for uptake into plants and soil invertebrates. The water-soluble fraction also provides a more realistic comparison to the toxicity tests used to derive numerical standards for the protection of soil invertebrates and plants because most of the available toxicological literature involves the spiking of a particular soil or growing medium with metal salt solutions.

3.2.2.2 Geochemical Conceptual Model and Leach Testing

Appendix A-3 provides a geochemical conceptual model based on the initial geochemical characterization work to identify the main factors that could influence the rate of copper leaching from tailings. The initial geochemical conceptual model identified that the potential for acid rock drainage potential would be negligible owing to low sulphur content (0.1 to 0.3%) and high buffering potential by the carbonate in the tailings. The potential for copper leaching would be low because copper was largely associated with the non-sulphide portion of the tailings, which is relatively insoluble and therefore would have a low potential for leaching under both subaerial and subaqueous conditions. Vanadium had a relatively low hazard based on the bulk soil concentrations (Section 3.2.1) and was not explicitly examined by SRK Consulting (SRK) as part the geochemical assessments.

Appendix A-2 provides data (from SRK) from a 90week kinetic test to confirm the hypotheses from the initial geochemical characterization. Leaching testing was conducted with representative samples of both the ‘grey’ and ‘magnetite’ tailings to simulate leachability under terrestrial conditions. Two different types of leaching tests were conducted:

- A humidity test where bulk soils are retained in a sealed chamber and a small amount of water added once per week to flush out the copper that had dissociated from tailings in the previous six days
- A column test where water is slowly trickled into the tailings on a continuous basis

Copper leaching rates at stability ranged from 0.00043 to 0.0019 mg copper per kilogram (Cu/kg) tailings per week. These solute release rates represent a worst-case concentration of the porewater copper concentration in tailings which would be accessed by plant roots because these concentrations would be reduced by transformation, sorption, and attenuation processes that will occur over time as the physical attributes of the soil improve. Appendix A-2 concluded that the leach tests confirmed their finding that the tailings are not acid-generating and will remain neutral or alkaline. Appendix A-2 also concluded that the presence of dissolved organic carbon and the formation of secondary copper minerals (i.e., solubility limits) would tend to keep copper at a ceiling concentration (i.e., copper concentrations in porewater will not increase proportionally to the soil concentration). The solubility of the copper complexes would be limited once porewater had migrated further than half a meter (i.e., copper will form precipitates and have reduced bioavailability to plants and soil invertebrates).

The underlying geochemical characterization was also considered for other exposure scenarios:

- SRK (2015) concluded that the potential for copper to leach from tailings that are subaqueous (i.e., at the bottom of Polley or Quensel Lake) was limited, based on the mineralogical characterization and sequential extractions. This conclusion focused on two specific geochemical mechanisms that could contribute to copper release. First, copper that binds to sulphides in sediment would be unlikely to be released because water saturation would be expected to inhibit oxidation of sulphides. This mechanism is the basis for AVS-SEM measurements (see Section 4.6.1). Second, iron oxyhydroxides tend to sorb divalent metals such as copper, but could then re-release the copper if the iron oxyhydroxide dissolved under reducing conditions. SRK concluded that iron oxyhydroxides were a minor component of the tailings geochemistry, and therefore, the presence of reducing conditions would be unlikely to result in the remobilization of large amounts of copper.

- Appendix A-3 provided an update to the geochemical conceptual model that reviewed several factors that were contributing to elevated copper concentrations in standing water captured in a few depressions representing a small fraction of the total impacted site area. The geochemical conceptual model was informative to the risk assessment in that it showed that higher than expected concentrations of copper, noted by MPMC and BC MoE in ditch water and pooled water in the Polley Flats area of upper Hazeltine corridor, is not in fact representing dissolved copper, but copper that is complexed with organic carbon and not readily bioavailable. The geochemical conceptual model was developed to explain current copper concentrations and understand expected seasonal fluctuations in copper concentrations in temporary ditches or pools of standing water. However, the geochemical conceptual model is not directly relevant to the risk assessment because it is not reflective of the majority of the site and it was concluded in the DSI that soil leaching into groundwater reaching habitats used by aquatic life was not a relevant pathway.

3.2.2.3 Physiological Based Extraction Testing (PBET)

The physiologically based extraction test (PBET) is a measure of the proportion of total metals in soil that are leachable (i.e., bioaccessible) in the mammalian gut. In brief, the method involves treating soils with solutions that simulate the acidic nature and processes of a typical mammalian gastrointestinal tract and measuring the total amount of metals that are released from the soil over the various steps of the extraction test. The method was initially developed to improve human health risk assessment models by accounting for site and sample-specific differences in the amount of lead (and later, arsenic) that would be absorbed by humans as a result of soil ingestion The PBET test was validated for this application by comparing the results of the PBET tests to the concentrations absorbed by pigs during feeding trials (Ruby et al. 1999). PBET is also a relatively common approach for improving food chain models for wildlife receptors and has been accepted by the MoE as part of ecological risk assessments at other sites for copper (i.e., Britannia Mine). PBET data was incorporated as a quantitative element of the wildlife food chain model (see Section 3.4.2).

However, the PBET data also provides a qualitative line of evidence about the overall bioavailability of copper and vanadium in a manner similar to the water soluble fraction and leach test data. PBET analyses were conducted on a total of seven representative samples of tailings, as well as an additional nine samples from rehabilitated areas (see Section 3.2.1). A detailed summary of method and results for the individual samples is provided in Appendix B-7, but overall, the PBET tests had an average bioaccessibility of 39% for copper and 3% for vanadium. Note that the PBET test involves extraction with an acidic solution with a pH of 1.5, and therefore, the observation regarding the bioaccessbility of copper is consistent with the conceptual geochemical model as well as the findings from the leaching test and the water soluble fraction measurements (see Section 3.2.2.1) which highlight that copper is not bioavailable unless in an acidic environment.

3.2.3 Soil Physical Characteristics and Nutrient Parameters

The soil ecosystem within the Hazeltine corridor was reduced to an early primary successional state dominated by exposed tailings and glacial sediments, including tills, and glaciofluvial and glaciolacustrine sediments. These materials are low in nutrient content and organic matter and most have a fine-grained, homogenous texture that can limit the flow of water and oxygen as well as restrict plant rooting. These physical characteristics are not conducive to diverse plant growth and could potentially retard the re-establishment a successional community. Re-establishment of an adequate soil biological community is needed to support the development of a more natural physical soil structure with adequate nutrients that can support microbes, plants and soil invertebrates (Coleman 2008). Two specific metrics were examined in this line of evidence:

- Physical characteristics of tailings — particle size and bulk density of tailings
- Plant available nutrients and total organic carbon — as measured by available nitrate as N, available potassium, available phosphorus, available sulphur and total organic carbon

3.2.3.1 Physical Characteristics of Tailings

Twenty tailings samples were submitted for grain size analysis and bulk density. A description of sampling methods and quality assurance and quality control methods are described in the original report (MPMC 2015) including sample locations, sample technique, laboratory analysis methods, original certificates of analysis, chain of custody forms, summary of quality assurance/quality control procedures (e.g., blind field duplicates, field replicates and laboratory duplicates) as well as supplemental information such as soil descriptions. Golder has reviewed these original reports for quality assurance/quality control practices were appropriate, and based on this review found that data were considered reliable for the purposes of the risk assessment. Key findings were:

- The average particle size results for tailings samples collected in 2014 were 49% sand, 41% silt, 9.6% clay and 1.7% gravel (Table 12). The physical structure of the tailings were observed to be different than native background soils, which were described as a silt with variable content of sand and gravel.
- Bulk density results (MPMC 2015) are presented in Figure 6. The average density of tailings was estimated to be 1.54 g/cm^3 assuming an average particle density of 2.65 g/cm^3.

Table 12: Summary of Tailings Soil Texture

Parameter	Tailings (n=20)		
	Minimum	**Average**	**Maximum**
Sand (%)	0.6	49.0	92.7
Silt (%)	6.8	41.4	80.2
Clay (%)	0.5	9.6	31.5
Gravel (%)	0.1	1.7	17.1

Data are from Table M in Section 5.5.1 of the soil quality impact assessment conducted by SNC-Lavalin (MPMC 2015 Appendix D).

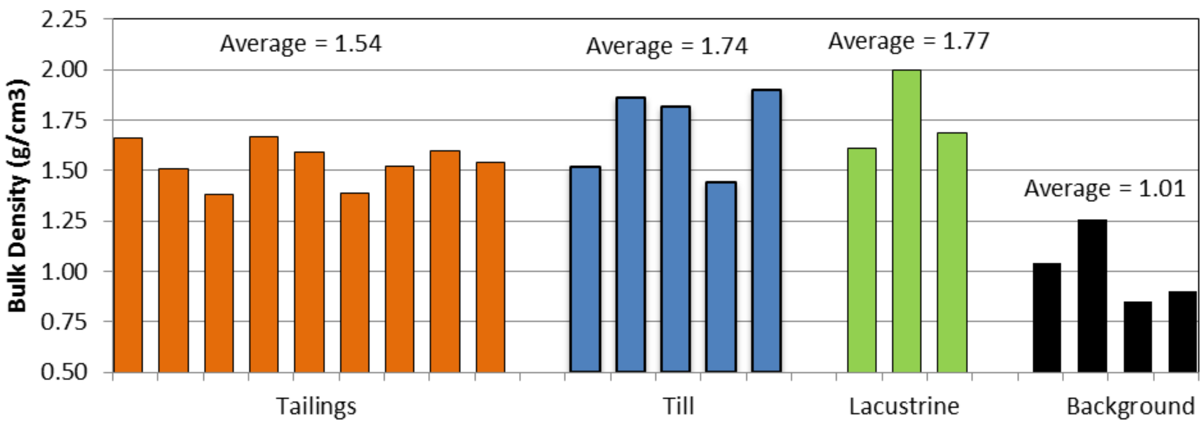

Figure 6: Summary of Bulk Density Measurements (From Figure A, Section 5.5.2 of Appendix D of MPMC 2015)

A potential mechanism of effect on plants relates to how roots interact with the pores in soils: root growth can become restricted if the average pore radius becomes significantly less than the diameter of the growing roots (Daddow and Warington 1983). Soil particle size has a direct effect on the average pore radius, while bulk density has an indirect effect on porosity. Threshold ranges for where elevated bulk density can be a contributing factor to poor plan growth from the literature include

- Approximately 1.4 to 1.6 gram per cubic centimeter (g/cm^3) dependent on soil texture (Brady and Weil 1996; Bulmer and Krzic 2003).
- Daddow and Warrington (1983) concluded that plant growth could be restricted at a bulk density as low as 1.4 g/cm^3 for silty clay loams and up to 1.75 g/cm^3 for loamy sands and sands.
- Bulk densities of greater than 1.7 g/cm^3 for mine soils were described as severely compacted and unable to sustain a healthy plant community (Sheoran et al. 2010).

The available soil density data were considered in light of texture-specific thresholds. The bulk density value that would likely result in restriction of root growth is estimated to be between 1.55 g/cm^3 and 1.60 g/cm^3 (MPMC 2015). The bulk density of till and lacustrine deposits is substantially higher than this threshold range, and therefore, root growth would be expected to be restricted. Conversely, background soils have an average bulk density that is substantially lower than this value, and therefore, root growth is not likely to be restricted by the pore radius. The bulk density of tailings is only marginally less than the threshold range, whereas the tailings are more similar to till than the background soils.

Golder concludes that the bulk density of the till samples is likely a physical stressor that would limit root growth. These physical stressors were also identified as the major factor limiting natural revegetation during the forest surveys and became the primary focus for the rehabilitation process (i.e., mixing and scarification is primarily intended to open up the soils to facilitate root growth as well as water and oxygen flow). The conclusion that the bulk density of the tailings is likely problematic to the establishment of a healthy plant community has a relatively high degree of certainty given that bulk density is a reproducible and quantifiable soil variable that has been compared to a threshold value consistent with multiple literature sources and which has a plausible mechanistic basis.

3.2.3.2 Plant Available Nutrients and Total Organic Carbon

Plant available nutrients and total organic carbon have been measured in a large number of samples during the course of the investigation. There are sufficient samples to provide spatial coverage for the variety of materials present (e.g., tailings, local reference organic soils, local reference mineral soils and soils within the rehabilitation zones; see Section 3.2.1.2). A description of the sampling and quality assurance and quality control methods are provided in the original reports (MPMC 2015; Appendix A; Appendix G-1). Golder has reviewed these original reports and consider the quality assurance/quality control practices were appropriate, and that data were considered reliable for the purposes of the risk assessment. A summary of the total plant available nutrients (i.e., the sum of available sulphate, phosphate, potassium and nitrate) is provided in Figure 7 along with the total organic carbon content. The individual plant available nutrients (sulphate, phosphate, potassium and nitrate) were also correlated to the total organic carbon content (Figure 8).

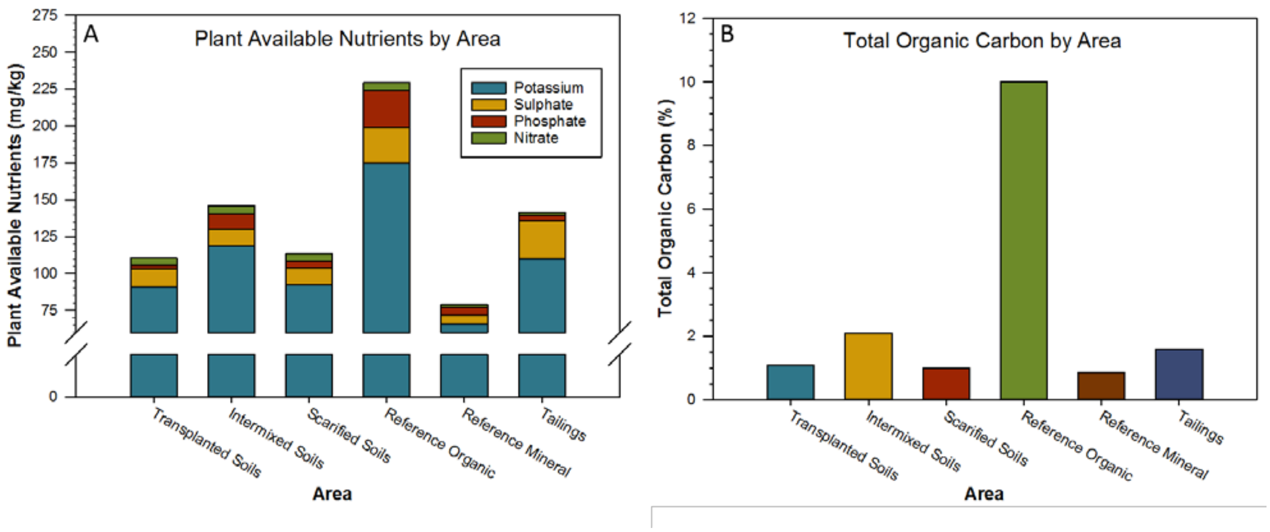

Figure 7: Plant Available Nutrients by Area (A) and Total Organic Carbon by Area (B)

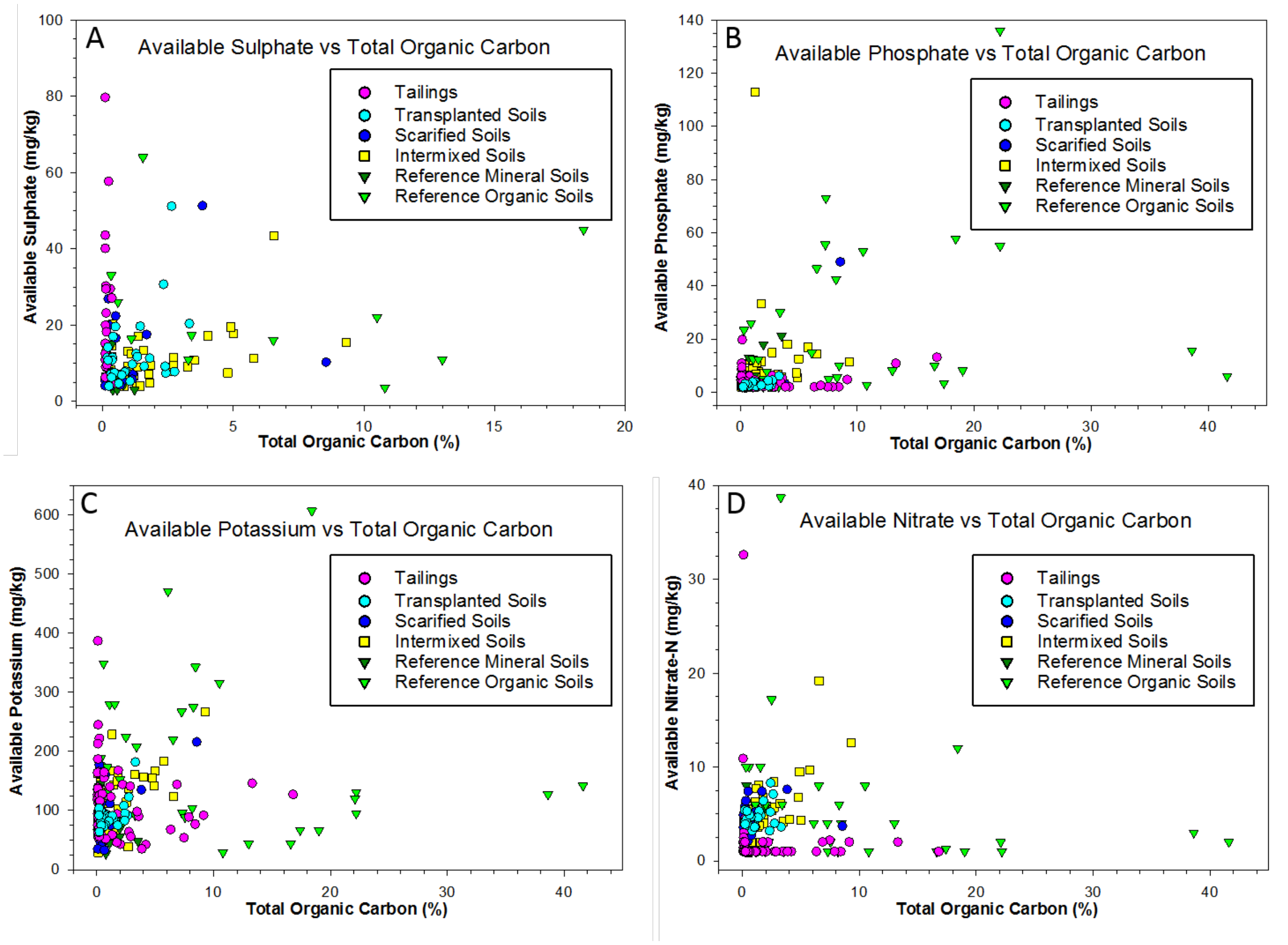

Figure 8: The Relationship Between Available Sulphate (A), Available Phosphate (B), Available Potassium (C) and Available Nitrate (D) and Total Organic Carbon for Samples within the Different Areas

Figure 7 and Figure 8 show a relatively high degree of variability in the individual nutrients amongst a soil type, but a general pattern where the total available nutrients and total organic carbon was lowest in a reference mineral soil and highest in reference organic soils. Tailings (and the three rehabilitation treatments involving tailings) also appear to have low total available nutrients and organic carbon that is more similar to the reference mineral soil.

Golder conducted a review to determine if there are threshold values for plant available nutrients in the literature. Multiple citations were identified indicating that tailings typically have limited amounts of necessary macronutrients until an active biological community capable of facilitating nutrient cycling has been re-established. Unsurprisingly, most studies involving revegetation success and plant growth on mine tailings tended to have very high concentrations of multiple metals, often on acid-generating wastes that would mobilize metals. These types of sites are not consistent with the current study area. However, one study was identified that highlights the influence of physical characteristics of a material on plant growth in the absence of metals or other major confounding toxicological factors. In brief, Reid and Naeth (2005a,b) conducted greenhouse and field trials on revegetation of dewatered kimberlite mine tailings with seven plant species native to the subarctic:

- The unamended kimberlite tailings had a circumneutral pH (pH = 8), a total organic carbon content of 0.2% and a sand fraction of 83%. These values are comparable to those observed in Mount Polley tailings. A comparison of the approximate plant available nutrients showed that plant available potassium was generally consistent (200 mg/kg in kimberlite versus 125 mg/kg in tailings), consistent for phosphate (10 mg/kg in both), higher for sulphate (285 mg/kg in kimberlite versus 20 mg/kg in tailings), and slightly lower for nitrate (2 mg/kg versus 5 mg/kg in tailings).
- A number of different amendments were tested in the greenhouse trial (Reid and Naeth 2005a). Peat and lake sediment were added to improve the physical characteristics of the kimberlite. Various sludges and fertilizers were added to improve the amount of plant available nutrients. Negative control performance (kimberlite) showed successful plant growth, but the largest improvement in plant growth occurred when both the physical and nutrient deficiencies were addressed. The field trial focused on the same amendments and species (Reid and Naeth 2005b). The negative control (kimberlite) showed successful plant growth that out-performed some of the amendments (i.e., addition of calcium-based fertilizer). As in the greenhouse study, the largest improvement occurred when both the physical and nutrient deficiencies were addressed.

3.2.3.3 Decision Criteria

Golder concludes that the appropriate decision criterion for evaluating total plant available nutrients is the qualitative comparison of tailings to the natural mineral and natural organic soils. On a qualitative basis, the tailings have not yet re-established levels of plant available nutrients that are consistent with the natural organic soils. There is qualitative evidence that mounding and other rehabilitation treatments are beneficial in this regard. The site-specific observations are consistent with the greenhouse and field trials conducted by Reid and Naeth (2005a,b) which demonstrated that structural and nutrient limitations in a material similar to the tailings at the site were able to exert negative effects on plant establishment and growth in the absence of any metal contamination. Reid and Naeth (2005a,b) also showed over a multi-year field trial that native plants were able to re-establish themselves at a higher success rate once the material's physical deficiencies had been mitigated.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Soil structure and nutrients	Protect microbes, soil invertebrates and plants from potentially hazardous exposure from metals via direct contact with soil	Measure plant available nutrient concentrations and total organic carbon in soil samples	Qualitative comparison of impacted areas to background.	Site wide	Tailings have lower plant available nutrient concentrations than background areas.
					Tailings have lower organic carbon content than background areas

3.3 Effects-Based LOE: Lower Trophic Level Receptors

3.3.1 Soil Invertebrate Tissue Chemistry

Measuring metal concentrations in field-collected soil invertebrates provides a direct measurement of bioavailability under actual exposure conditions. Tissue chemistry can be used as a line of evidence for looking at risks to soil invertebrates in two ways:

- The concentrations of copper and vanadium in tissue samples from exposed areas can be compared to the concentrations measured in reference areas.
- The concentrations of copper and vanadium can be compared to critical body residues from the scientific literature. A critical body residue is derived from toxicity tests, but calculates an LD50 (i.e., tissue concentration that results in 50% effect) instead of an LC50 (soil concentration that causes a 50% effect)[18].

The preliminary weight of evidence framework (Appendix G-2) also proposed a line of evidence where tissue chemistry would be used as a line of evidence to evaluate risks to wildlife receptors. This approach was superseded by using the tissue chemistry data as an input variable in a mechanistic food chain model (see Section 3.4.2).

3.3.1.1 Study Design and Methodology

Detailed information on the types and locations of collected soil invertebrates can be found in Appendix E-2, but in brief:

- Samples of soil invertebrates (ants, beetles, spiders, worms, and slugs) were collected in summer 2015 and summer 2016 from the Hazeltine corridor. The majority of soil invertebrate samples were collected from the halo area, although all areas along the Hazeltine corridor were searched. Samples were also collected from background forest areas that were at minimum 20 m beyond the extent of visual tailings deposition.
- Soil invertebrates of the same type from a given location were composited where necessary to provide sufficient sample mass[19]. There were no composites made by combining different taxonomic groups. All samples were washed prior to analysis. Worm samples collected in 2016 were depurated, but worm samples collected in 2015 were not. A co-located surface soil sample was also collected at each tissue sampling location.
- Summary statistics (minimum, maximum, mean) of soil invertebrate tissue metal results were calculated and plotted to visually examine differences between exposed and reference areas for different taxonomic groups. A statistical analysis was not conducted because the data sets were not of sufficient size to perform meaningful non-parametric tests.

[18] There are multiple ways to express toxicity tests (e.g., NOEC, LOEC, ECx). The use of the LD50 and LC50 terms is meant to provide a plain language description of the difference between a critical body residue and a typical toxicity test result.

[19] The exception to compositing was for worms. Worms had sufficient mass such that an individual worm could be submitted as a sample.

- Bioaccumulation factors were also calculated for each tissue sample using the co-located soil concentration (normalized to bulk density) to visualize the relationship. Bioaccumulation factors were plotted against soil concentrations to examine if there was a concentration-dependent relationship of metal accumulation in soil invertebrates. These relationships were also considered in the decision-making regarding the parameterization of the food chain model (Appendix J).

3.3.1.2 Relative Differences between Sampling Areas and Uptake

A detailed presentation of the individual sample concentrations, tabular summaries, discussion of quality assurance/quality control measures and the original certificates of analysis is provided in Appendix E-2. Data were considered reliable and suitable for consideration in the risk assessment. A variety of decision criteria for this line of evidence were considered, but ultimately, the limited number of samples means that it is problematic to make a quantitative comparisons between years or areas for any given sample type. Golder concluded that the limited number of samples meant that it was not defensible to use statistical significance as the decision criterion. The range of observed tissue concentrations and the relationships between soil concentrations and bioaccumulation factors are provided in Figure 9 and Figure 10. Information about a 20% increase relative to background, and the upper limit of the normal range of background variability (i.e., average background concentration plus two standard deviations; Kilgour et al. 1998, Barrett et al. 2015) is provided in Table 13 for additional context, but ultimately Golder is not drawing a conclusion about whether adverse effects to soil invertebrate populations are likely based on this line of evidence because of the high uncertainty in the tissue concentration data. The line of evidence is still important to provide context to the overall conclusions.

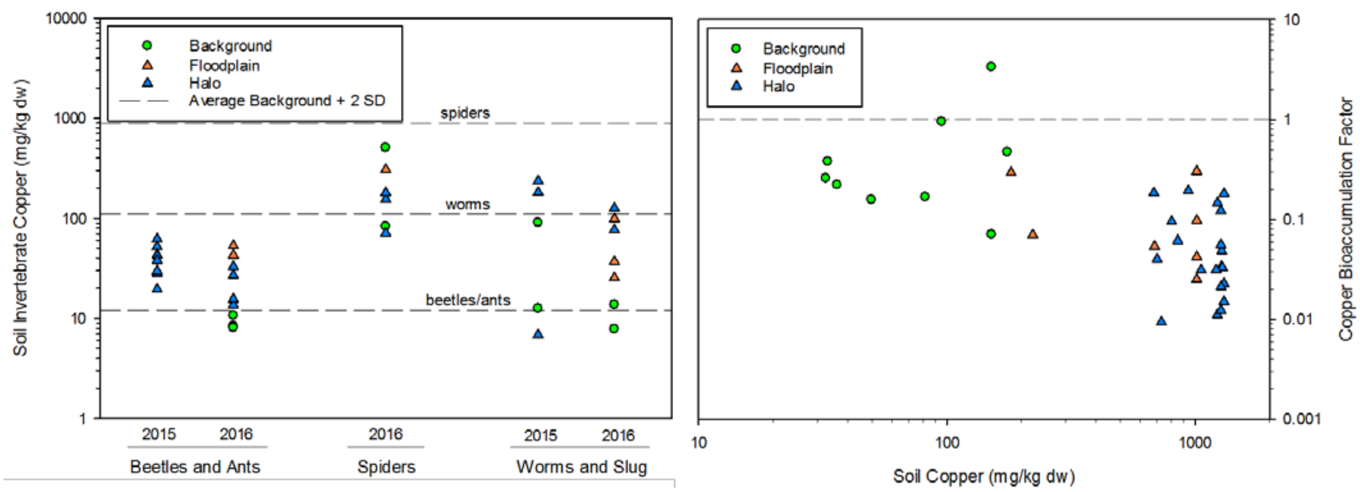

Figure 9: Soil Invertebrate Tissue Concentrations and Bioaccumulation Factors for Copper

Note: SD = Standard Deviation.

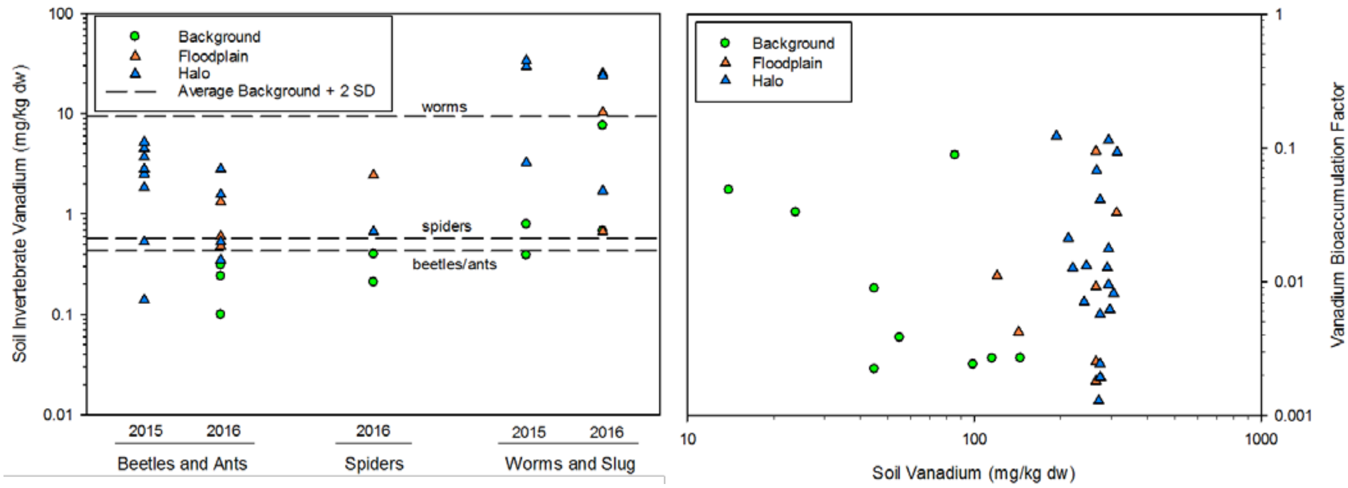

Figure 10: Soil Invertebrate Tissue Concentrations and Bioaccumulation Factors for Vanadium

Note: SD = Standard Deviation.

Table 13: Comparison of Copper and Vanadium Soil Invertebrate Tissue Concentrations.

Parameter	Invertebrate Type	Hazeltine corridor		Background		Context	
		n	Average	n	Average	Average + 2SD	Average + 20%
Copper	Beetles/Ants	15	**34**	3	9.1	12	10.9
	Spider	4	178	2	296	896	355
	Worm/Slug	8	99	4	31	111	37.2
Vanadium	Beetles/Ants	15	**1.9**	3	0.22	0.43	0.26
	Spider	4	**8.1**	2	0.31	0.57	0.37
	Worm/Slug	8	**16**	4	2.4	9.4	2.9

Concentrations presented are in milligrams per kilogram dry weight.

Bold indicates the average concentration in samples from the Hazeltine corridor are greater than the average background concentration plus 2 standard deviations (SD).

The key qualitative observations were:

- The range of copper and vanadium concentrations in soil invertebrate samples from the halo and floodplain tended to overlap with the range of concentrations observed in background, despite substantially higher soil concentrations in the exposed areas.
- Variability of tissue concentrations for a given metal and invertebrate type was relatively high for copper, but spiders appear to have a higher tissue concentrations in both exposed and background areas. A concentration-dependent bioaccumulation factor may be present (see Appendix J for further evaluation of this pattern for specific taxonomic groups). Broadly, this pattern suggests that some degree of bioaccumulation may occur in terrestrial food webs, but the low bioaccumulation factors suggest that risks associated with bioaccumulation from tailings may be lower than what would be expected based on consideration of the bulk soil concentrations alone.

- Variability in tissue concentration of vanadium was also high, but there were no apparent differences between area or taxonomic groups. Broadly, this pattern suggests that both bioaccumulation (and by extension, risk to soil invertebrates) is likely less for vanadium than for copper.

3.3.1.3 Critical Body Residue

Golder conducted a brief review of the literature to determine if there were adequate data to support the derivation of a critical body residue value (i.e., a tissue concentration that corresponds to a toxicological effect). No data were identified for vanadium for soil invertebrate taxa. Data for copper were also not identified for any taxonomic group except earthworms. There was considerable variability in the literature with respect to an internal tissue concentration in earthworms that corresponded to a threshold level of effects.

- Streit (1984) reported that mortality to the earthworm *Octolasium cyaneum* was evident at tissue concentrations greater than 100 – 120 mg/kg dry weight in a laboratory-spiked soil study.
- Ma (2005) suggested a sublethal threshold of approximately 60 mg/kg dry weight based on a spiked-soil study with *Eisenia foetida*, but concluded that a critical body residue approach was subject to considerable uncertainty in terms of applying the threshold values to a management decision. Ma (2005) concluded that a critical body residue approach may still have value if combined with other lines of evidence as part of a bioavailability-based investigation.
- Garcia-Gomez et al. (2014) concluded that critical body residues for copper for *E. foetida* did not correspond to either a concentration-response relationship with soil concentrations, or to a toxicological dose-response relationship for survival or growth. Tissue concentrations greater than 220 mg Cu/kg dry weight corresponded to greater than a 50% reduction in both survival and the growth of surviving earthworms, but a dose-response relationship was not evident below that upper limit.

Golder concluded that the available literature information was only sufficient to establish a preliminary critical body residue value. The average copper concentration in the small number of worm samples available from the site is greater than the lowest threshold value obtained from the literature. Golder is not prepared to conclude that there are risks to soil invertebrates associated with copper based on this limited data, especially because copper is an essential element and organisms have physiological mechanisms to maximize copper uptake when copper is abundant and to moderate copper uptake when it is not. The magnitude of bioaccumulation is also influenced by site-specific factors that can limit bioavailability: this interaction between geochemical and physiological variables was evident in copper bioaccumulation testing using springtails (*Folsomia candida*) (Vijver et al. 2001). Vijver et al. (2001) found that springtails maintained a constant body residue concentration irrespective of the soil or porewater concentration. There is also considerable variability amongst the three studies in terms of selecting a defensible threshold tissue concentration for earthworms. As a result, Golder concludes that this line of evidence is at best a qualitative indicator based on the data currently available.

3.3.1.4 Influence of Physical Stressors

From a practical sense, the difficulties in obtaining a sufficiently large sample size also provides insight about the relative influence of physical versus chemical stressors. It was difficult to find sufficient invertebrates in the impacted areas for sampling, and therefore, Golder concludes that the soil invertebrate community has not yet recovered from the initial impact of the tailings deposition. Specific thresholds for when physical properties such as bulk density or total organic carbon in tailings would limit recolonization by different types of soil invertebrates were not identified in the literature. One possible threshold relates to an examination of potential artificial and negative control soils for earthworm toxicity testing (Environment Canada 2004). Bulk densities in the candidate soils ranged from 0.5 to 1 g/cm3, which is substantially more porous that the density measured in tailings (density: 1.5 g/cm3). Golder concludes that the physical nature of the tailings would likely be a barrier to successful recolonization by earthworms.

3.3.1.5 Risk Conclusion and Uncertainty Analysis

For the reasons described above, Golder is placing relatively low weight on tissue chemistry in terms of predicting risks to the soil invertebrate community. Although direct measurement of tissue concentrations provides the most realistic measure of bioavailability, there were relatively few samples available. The data that were available indicate that metal uptake was higher in the tailings area than in the reference area, however, the ecological relevance of that change in tissue chemistry was not clear because experimental or field-based measures of effect that can be related to the chemistry data were not available. The preliminary critical body residue value suggests that copper bioaccumulation could contribute to some level of biological effects in earthworms, but again, the low sample size made it difficult to determine a defensible estimate of the average concentration in the earthworm population. There was also the possibility that some tissue concentrations were biased high because of the presence of soil particles (despite the efforts to wash the samples). This bias was most likely present in the undepurated earthworm samples collected in 2015—these two samples had the highest measured concentrations of copper and vanadium but were retained in the data set as a conservative measure to avoid reducing the sample size even further.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Metal concentrations in tissues	Protect soil invertebrates and plants (collectively, the lower trophic levels) from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Measure metal concentrations in field-collected soil invertebrate tissues.	Qualitative comparison of impacted areas to background.	Site wide	Soil invertebrates in impacted areas have higher Cu tissue concentrations than background

3.3.2 Plant Tissue Chemistry

Measuring metal concentrations in field-collected plants provides a direct measurement of bioavailability under actual exposure conditions. Plant tissue chemistry can be used as a line of evidence for looking at risks to plants by comparing the concentrations of copper and vanadium in tissue samples from exposed areas to the concentrations measured in reference areas. Critical body residues (which were considered for soil invertebrates) do not apply to plants. The preliminary weight of evidence framework (Appendix G-2) also proposed a line of evidence where tissue chemistry would be used as a line of evidence to evaluate risks to wildlife receptors. This approach was superseded by using the tissue chemistry data as an input variable in a mechanistic food chain model (see Section 3.4.2).

3.3.2.1 Study Design and Methodology

Detailed information on the types and locations of collected plants can be found in Appendix E-1, but in brief:

- The Hazeltine corridor and its surrounding background area was searched in the summer months of 2015 and 2016 for plants that would be consumed by herbivores. The number of species that would likely be consumed by herbivores was limited. Conifer (spruce) and berry (thimbleberry, blueberry, Devil's club, highbush cranberry and bunch berry) samples were available from the halo zone. Grasses (rye and barley) and edible leaf samples were available from the floodplain area in 2016 because of planting efforts by MPMC the previous year.
- A soil sample was collected from the base of each sampled plant. Tissue samples were rinsed with deionized water and blotted dry by the analytical laboratory prior to analysis.
- Copper concentrations in different plant tissues collected from the Mount Polley property were available from 1989, 1995, and 1996 prior to the start of mining operations.
- Summary statistics (minimum, maximum, mean) of plant tissue metal results were compared to background and pre-mining baseline (where available) sample results. Data from 2015 and 2016 were not statistically different, thus data were pooled between the two years. Individual plant samples collected from the floodplain and halo areas in 2015 and 2016 were plotted for visual comparison to background and baseline samples. Boxplots were created for each plant tissue type to compare concentrations across areas.

Table 14: Number of Each Type of Plant Tissue Sample Collected Along Hazeltine Channel

Tissue	Year	Halo	Floodplain	Background	Pre-mining Baseline (Cu only)
Berry	2015	11	-	6	-
	2016	21		10	
Shrub	2015	1	20	10	85
	2016	6	19	11	
Conifer	2015	7	-	3	11
	2016	19		12	
Grass	2015	-	11	-	10
	2016	2	19	11	

3.3.2.2 Relative Differences between Sampling Areas

A detailed presentation of the individual sample concentrations, tabular summaries, discussion of quality assurance/quality control measures and the original certificates of analysis is provided in Appendix E-1. Data were considered reliable and suitable for consideration in the risk assessment. The preliminary decision criterion for soil invertebrate tissue chemistry (Appendix F-2) focused on a 20% increase in the average tissue concentration as an indicator of increased risk. This was revisited. Golder concluded that it was more appropriate to conduct a statistical comparison between the different areas and tissue types because it was not possible to determine whether a 20% change in tissue concentration was ecologically relevant or not. Statistical analysis were conducted with support from a subject matter expert (Dr. Dennis Helsell). Key findings from Figure 11 and Figure 12 were:

- There were no differences in tissue metal concentrations[20] between 2015 and 2016 data sets; therefore, data from both years were pooled for subsequent statistical analysis. Not all tissue types were available from all three areas (floodplain, halo, background). Shrubs were the only tissue type that was present in both the floodplain and the halo area. Floodplain and halo data for shrubs were kept as separate populations for the following reasons:
  - Differences in exposure—plants growing in the halo are generally rooted in native soils underlying tailings while floodplain plants have been planted directly into the deposited tailings or rehabilitation materials, which contain varying levels of tailings.
  - Differences in age—plants growing in the halo were generally well established and growing for several years, while all plants in the floodplain were planted post-TSF embankment breach as part of rehabilitation efforts.
- Copper concentrations in plants associated with tailings were not consistently elevated, suggesting that bioaccumulation may be limited, or there are mechanisms that limit translocation of copper amongst plant tissues. Bioaccumulation varied with the plant type and were as follows:
  - Concentrations in grasses and shrubs were greater[21] in the floodplan than background and pre-mining baseline.
  - Concentrations in conifers were greater in the halo than the local background, but not greater than the pre-mining baseline.
  - Copper concentrations in berries from the halo were not greater than in background samples. Baseline data were not available for berries.
  - Copper concentrations in halo shrubs were within the range of background and baseline shrubs[22].
- Vanadium concentrations in all plant types associated with tailings were greater than the concentrations from the local background, suggesting that bioaccumulation is occurring. The only exception was vanadium concentrations in halo shrubs, which were within the range of background samples with the exception of a single outlier. No baseline data were available for vanadium.

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[20] All discussion about difference in tissue concentration relates to the presence of statistically significant ($p > 0.05$) differences.

[21] "Greater" in this context refers to a statistically significant increase in concentration in samples collected from the exposed areas relative to the reference or pre-mining baseline.

[22] A statistical comparison was not conducted as the sample size (n=7) was considered too small.

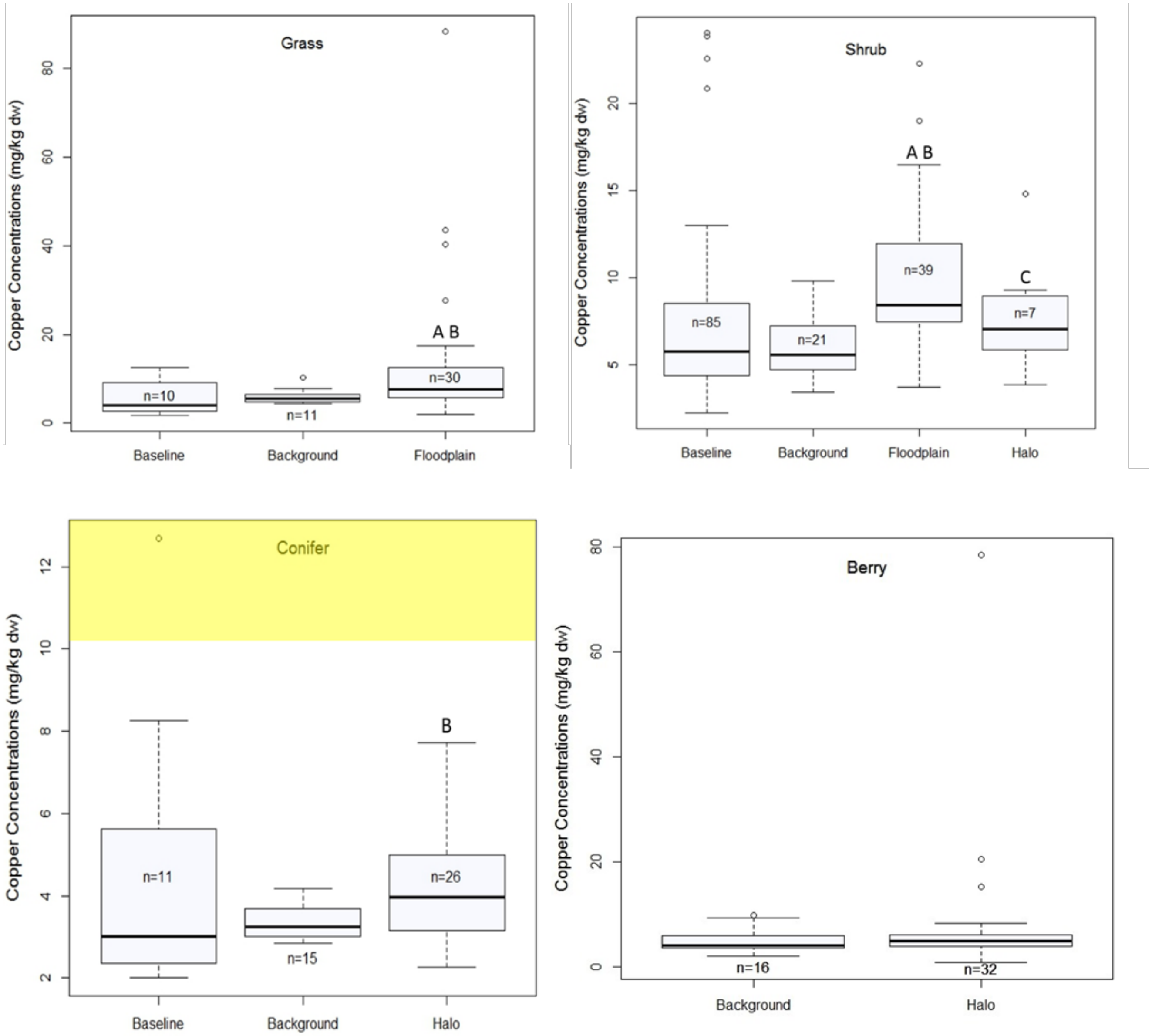

Figure 11: Copper Concentrations in Plants

Note: A = significant difference relative to pre-mining baseline; B = significant difference relative to local background; C = Insufficient data for statistical comparison to background or baseline.

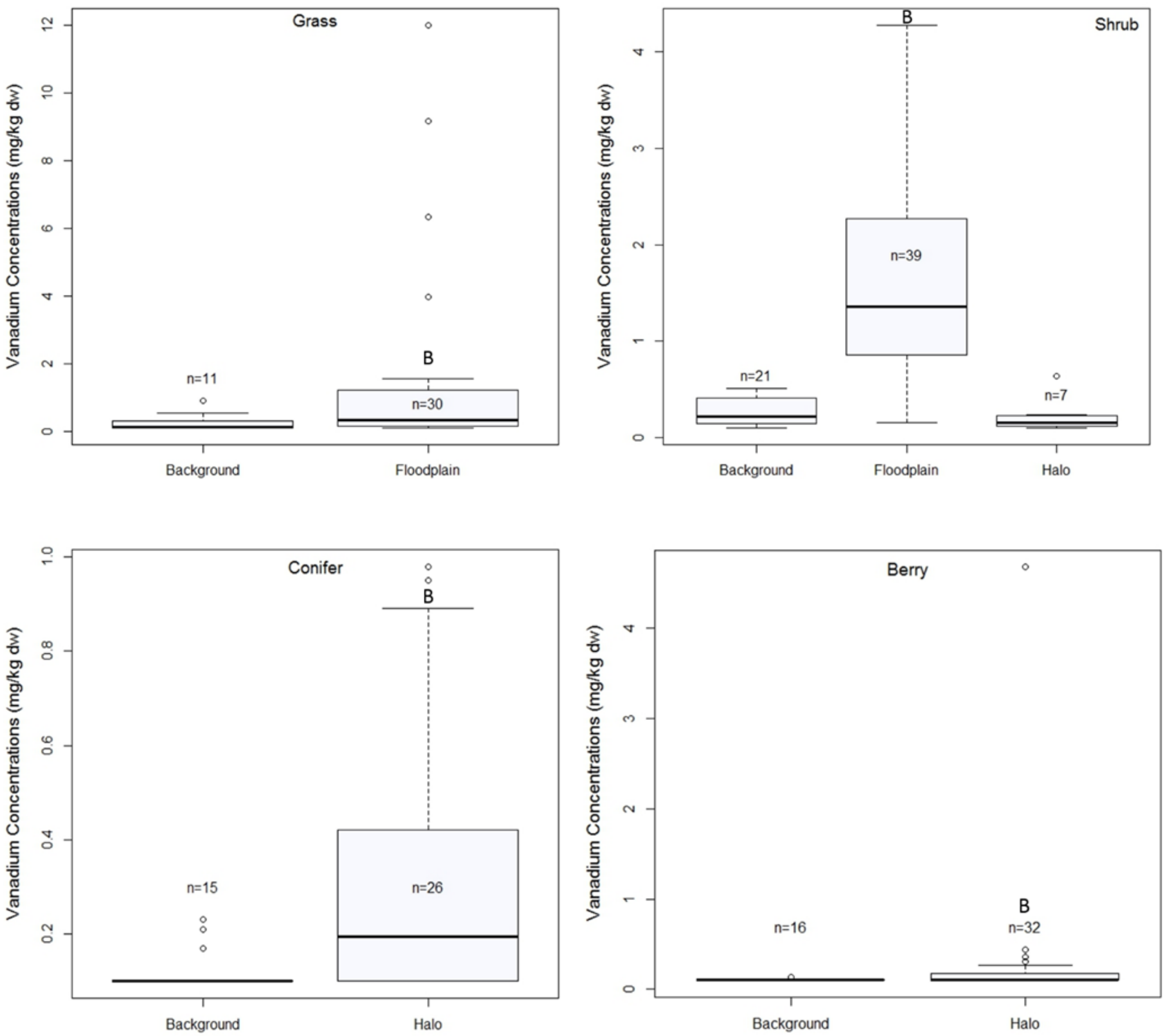

Figure 12: Vanadium Concentrations in Plants

Note: B = significant difference relative to local background.

3.3.2.3 Relative Uptake

Copper bioaccumulation factors in all plant types were inversely related to soil concentrations (Figure 13), consistent with their ability to regulate copper uptake (Emamverdian et al. 2015). Copper bioaccumulation factors were similar between background and exposed areas with the same range of soil concentrations, suggesting that copper bioavailability is similar between areas. Vanadium bioaccumulation factors were less dependent on soil concentrations.

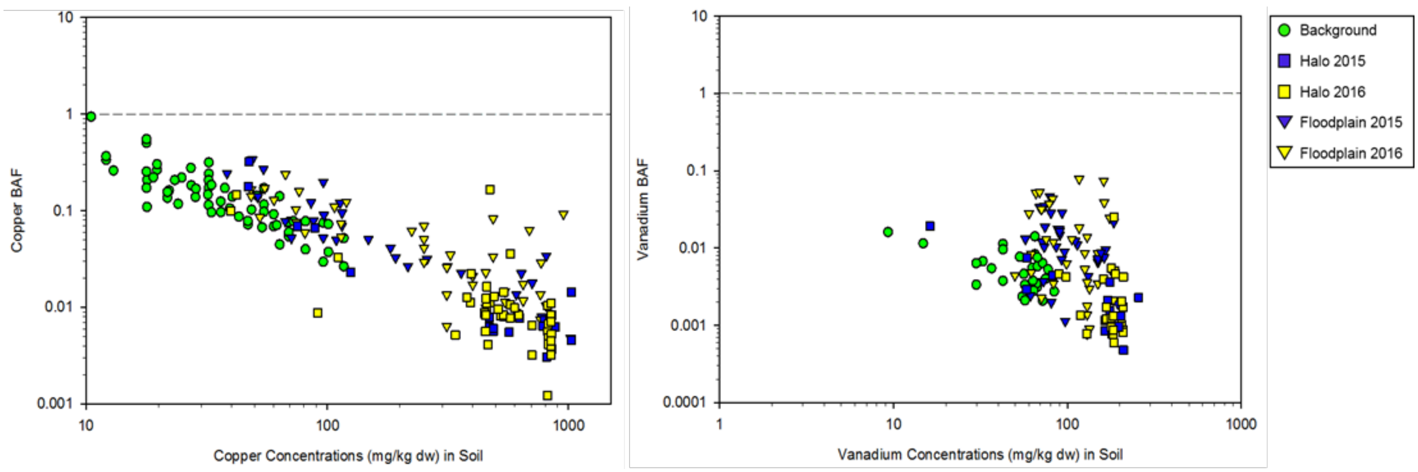

Figure 13: Copper and Vanadium Plant Bioaccumulation Factors

3.3.2.4 Adequate Copper Levels in Plants

Copper is essential for plant growth, but can be toxic at high concentrations. A “normal” concentration to meet nutritional needs of a plant varies by species and soil conditions. Hochmuth et al. (2015) concluded that the “normal” concentration for various vegetable plants was between 4 and 20 mg/kg dw. No data were identified for vanadium. The majority of tissue samples from the Hazeltine corridor had copper concentrations within the range considered normal (Figure 14).

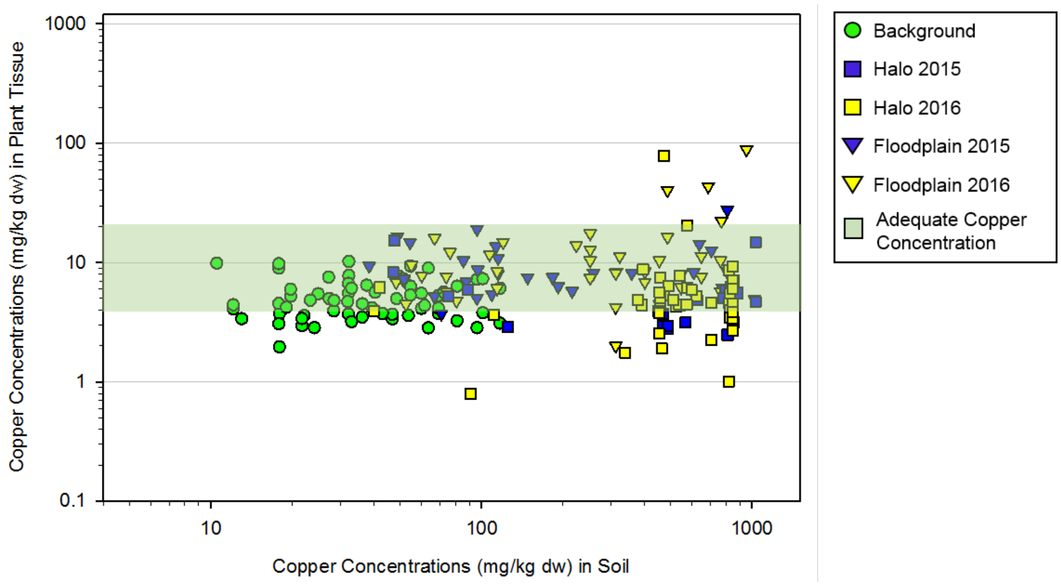

Figure 14: Copper Concentrations in Plant Tissue versus Soil

3.3.2.5 Relative Contribution of Physical and Chemical Stressors

Golder concluded that the physical nature of the tailings would likely limit revegetation (Section 3.2.3) in the absence of metals. Simard et al. (2003) also concluded that soils with very low organic carbon, nutrients and an impacted soil mycorrhizal community, and could affect regulation of nutrient uptake by plants.

3.3.2.6 Risk Conclusion and Uncertainty Analysis

Plant tissue chemistry data provides information about the overall risks to the plant community. Direct measurement of tissue concentrations provides the most realistic measure of bioavailability, and for plants, there was sufficient sample size to test for statistical differences.

- Risks to plants associated with copper based on this line of evidence are considered low. Copper had a bioaccumulation factor that was less than 1 and inversely proportional with soil concentrations. Not all plant types showed a statistically significant increase in copper concentration in soils impacted by tailings relative to reference, despite higher soil concentrations. The larger sample size also increases the reliability of using a ‘control chart’ to test the magnitude of difference beyond statistically significant differences; the ‘control chart’ approach showed that plant tissue concentrations were within the average plus two standard derivations for all plants except grasses and shrubs. The tissue concentrations were also frequently within the range of “normal” copper concentrations measured in agricultural crops.
- Risks to plants associated with vanadium based on this line of evidence are also considered low, but with higher uncertainty. There were no pre-mining baseline data available, and no literature was available to establish a ‘normal’ range of vanadium concentrations in plants. Vanadium bioaccumulation appears to be less dependent on soil concentration—either plants have a lower ability to regulate vanadium uptake, or plants tend not to accumulate vanadium.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Metal concentrations in tissues	Protect soil invertebrates and plants (collectively, the lower trophic levels) from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Measure metal concentrations in field-collected plant tissues.	Comparison of impacted areas to background.	Site wide	Plants in impacted areas have higher Cu and V tissue concentrations than background

3.3.3 Plant Toxicity Testing

Soil toxicity testing provides information about adverse effects of soil on surrogate plant species under controlled laboratory conditions. Toxicity testing was conducted with a variety of soil dilutions and included both unspiked and spiked (with copper) treatments. Overall, the data from the soil toxicity testing can be used to address two main questions:

- What is the effect of the tailings that have been placed along the Hazeltine corridor on seedling survival and growth?
- What is the relative influence of physical versus chemical stressors on the toxicological endpoints?

3.3.3.1 Toxicity Testing Methodology

Soil toxicity testing was conducted by Mr. Anthony Leung (MSc candidate) of the University of British Columbia (UBC) Department of Forest and Conservation Sciences, working under the supervision of Dr. Les Lavkulich and Dr. Suzanne Simard with input from Golder toxicologists. A detailed description of the study design, methods and results are provided in Appendix I, but in brief:

- Two bulk samples of tailings (referred to as 'sandy' tailings and 'silty' tailings)[23] collected from the Polley Flats were provided to UBC. Key differences in these bulk samples are summarized in Table 15, but overall, the tailings were largely similar with the exception of a small change in the total copper concentration and the difference in grain size distribution.

Table 15: Chemical and Physical Characteristics of Tailings Used in Plant Toxicity Testing

Parameter	Unit	Sandy Tailings	Silty Tailings
Copper	mg/kg	1130	805
Vanadium	mg/kg	187	202
Silt	%	14	47
Sand	%	66	44
Total Organic Carbon	%	0.6	0.4

[23] Note: "sandy" tailings refers to "magnetite" tailings and "silty" tailings refers to the "grey" tailings.

- Toxicity testing was conducted on dilutions of these two samples (created with a clean silica sand with a similar grain size as the original tailings) using lodgepole pine (*Pinus contorta*) and bluebunch wheatgrass (*Pseudoroegneria spicata*) seeds[24]. Toxicity tests were conducted under controlled conditions in general compliance with plant toxicity test methods described by Environment Canada and ASTM. A greenhouse soil was used as a negative control. All test replicates were supplemented with water with a nutrient solution to remove the likely confounding effect associated with low plant available nutrients.
- Test exposure duration was 42 days. Toxicological endpoints were seedling emergence, seedling survival, shoot and root length, and shoot and root dry weight. Weight and length provide growth endpoints, while seeding emergence and survival provide survival endpoints.

Golder has reviewed the soil toxicity data and concludes that it is defensible and appropriate for inclusion in the risk assessment.

3.3.3.2 *Effect of 100% Tailings*

The effects of the original, unamended tailings samples on the plant endpoints are provided in Table 16. Highlights include:

- There were no effects on seedling emergence or survival for either species in the two tailings samples.
- Lodgepole pine had more than a 20% reduction in root length and weight in both tailings relative to their respective negative controls, but no effects with respect to shoot length or weight.
- Bluebunch wheatgrass had more than a 20% reduction in all four growth endpoints in both tailings relative to the respective negative controls.

Table 16: Summary of Plant Toxicological Endpoints in Unamended Tailings Samples (mean ± SD)

Endpoint	Sandy Tailings	Silty Tailings	Sandy Control	Silty Control	Greenhouse Soil
Lodgepole pine					
Seedling emergence (%)	96 ± 9	100	96 ± 9	100	100
Seedling survival (%)	100	100	100	100	96 ± 9
Shoot length (cm)	3.1 ± 0.4	3.1 ± 0.4	4.1 ± 0.1	4.0 ± 0.4	3.8 ± 0.2
Shoot weight (g)	0.044 ± 0.013	0.054 ± 0.009	0.088 ± 0.012	0.067 ± 0.031	0.046 ± 0.013
Root length (cm)	**4.7 ± 1.7**	**3.1 ± 1.1**	13.6 ± 3.2	8.4 ± 4.8	6.3 ± 1.5
Root weight (g)	0.017 ± 0.006	0.021 ± 0.016	0.045 ± 0.009	0.020 ± 0.013	0.008 ±0.002
Bluebunch wheatgrass					
Seedling emergence (%)	96 ± 9	96 ± 9	96 ± 9	96 ± 9	80 ± 14

[24] A third plant species was also tested (wild willow, *Salix scouleri*); however, germination success was poor across all treatments including the controls, and the test was not completed. See Appendix I for more details.

Endpoint	Sandy Tailings	Silty Tailings	Sandy Control	Silty Control	Greenhouse Soil
Seedling survival (%)	95 ± 11	98 ± 4	88 ± 11	100	96 ± 9
Shoot length (cm)	**11.5 ± 2.7**	**13.1 ± 1.1**	22.5 ± 3.3	30.9 ± 2.9	27.3 ± 3.7
Shoot weight (g)	**0.050 ± 0.020**	**0.058 ± 0.030**	0.187 ± 0.056	0.340 ± 0.037	0.120 ± 0.059
Root length (cm)	**5.7 ± 2.3**	**7.6 ± 1.1**	23.8 ± 2.9	27.6 ± 1.9	13.8 ± 1.9
Root weight (g)	**0.022 ± 0.011**	**0.025 ± 0.012**	0.161 ± 0.055	0.241 ± 0.064	0.025 ± 0.011

Bold underlined indicates greater than a 20% reduction relative to the respective negative control.

3.3.3.3 Relative Contribution of Physical and Chemical Stressors

A dilution series of each tailings sample was created with its matching silica sand (i.e., 100, 50, 25, 10 and 0% tailings) and run in comparison with a silica sand series spiked with different concentrations of copper sulphate, a more bioavailable form of copper than what would be found in Mount Polley tailings. For example, the 100% sandy tailings had a copper concentration of 1,130 mg/kg, which means the 50% sandy tailings would have a nominal concentration of 566 mg/kg, 25% sandy tailings a nominal concentration of 284 mg/kg, and 10% sandy tailings a nominal concentration of 115 mg/kg. Identical concentration series were then created by spiking different concentrations of copper sulphite into the sandy control, the silty control, and the greenhouse soils. The objective of the study was to examine whether or not the toxicological endpoints in the five different concentration series share a similar concentration-response relationship. If they did, this would provide strong evidence that adverse effects are caused by the copper. If not, this would suggest that the copper concentrations in the 100% tailings were not the sole cause of the observed reductions in plant growth noted above in Table 16. The compiled results of the side-by-side spiked copper trial are provided in Figure 15 and Figure 16.

Copper concentrations were also measured in the substrates at the beginning of the toxicity test as well as in plant tissues at the completion of the toxicity test. Chemical analyses were conducted by UBC using a rapid wet digestion method described by Pequerul et al. (1993)[25] and the filtered extracts were analyzed using inductively coupled plasma-atomic emission spectroscopy. Relationships between copper concentrations in plants tissues and the exposure concentration in substrate are provided in Figure 17.

[25] The method involves extraction with nitric acid and hydrogen peroxide under heat, followed by further exposure to hydrochloric and nitric acid. It is not clear the degree to which this method is consistent with standard methods for soil and plant tissue analysis reported elsewhere in this risk assessment. Golder concludes that the data provides reasonable information about the relative copper concentrations in soil and plant samples for the toxicity testing, but it would be inappropriate to relate these data quantitatively to other lines of evidence given the differences in the chemical analysis method. A formal certificate of analysis was not available for review; data are assumed to be reliable as provided.

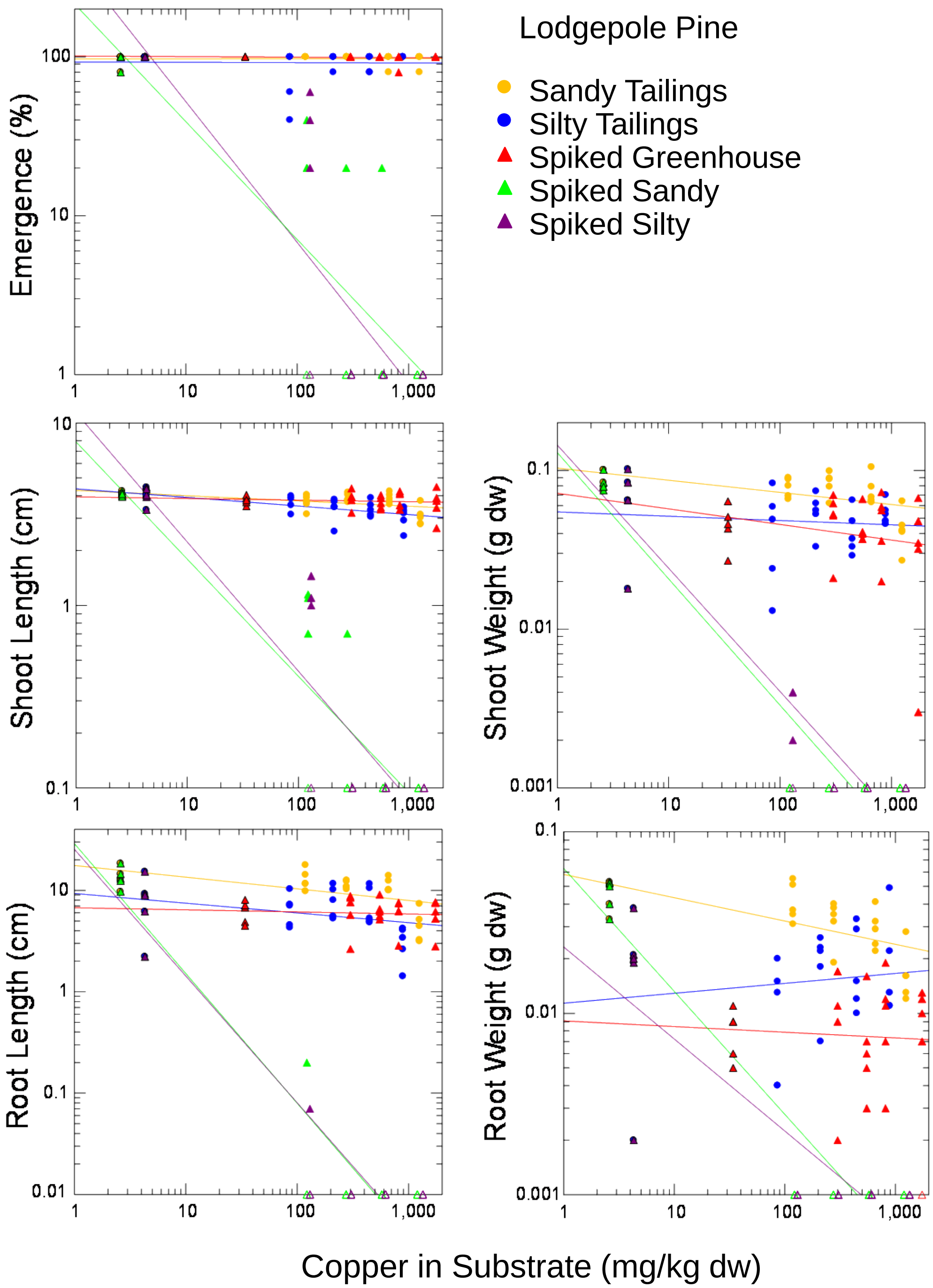

Figure 15: Summary of Concentration-Response Relationships for Spiked and Non-Spiked Copper and Various Toxicological Endpoints in Lodgepole Pine (Pinus contorta)

Note: circles = diluted tailings, triangles = copper sulphate-spiked substrate, black outlined symbols = controls (same sand or silt controls for diluted tailings and spike substrate), open symbols = response values adjusted from 0 to display on log scale (often multiple replicates). Lines are best-fit regression for visual comparison and colours correspond to substrate type. Results are from individual replicates (n=5 for controls, tailings, greenhouse soil; n=4 for spiked substrate). Measured concentration of copper in substrate for each treatment (n=1).

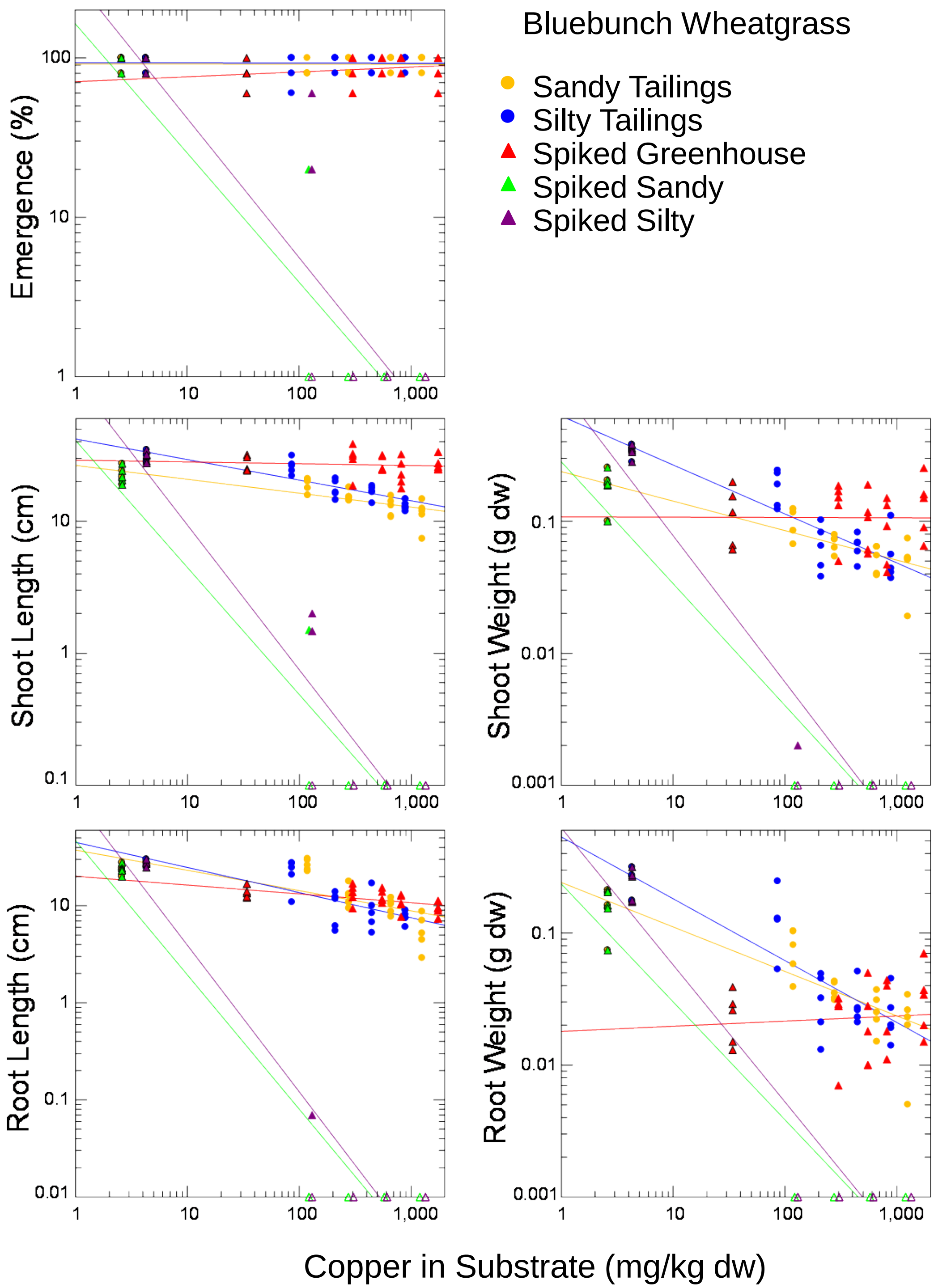

Figure 16: Summary of Concentration-Response Relationships for Spiked and Non-Spiked Copper and Various Toxicological Endpoints for Bluebunch Wheatgrass (Pseudoroegneria spicata)

Note: See Figure 15 notes.

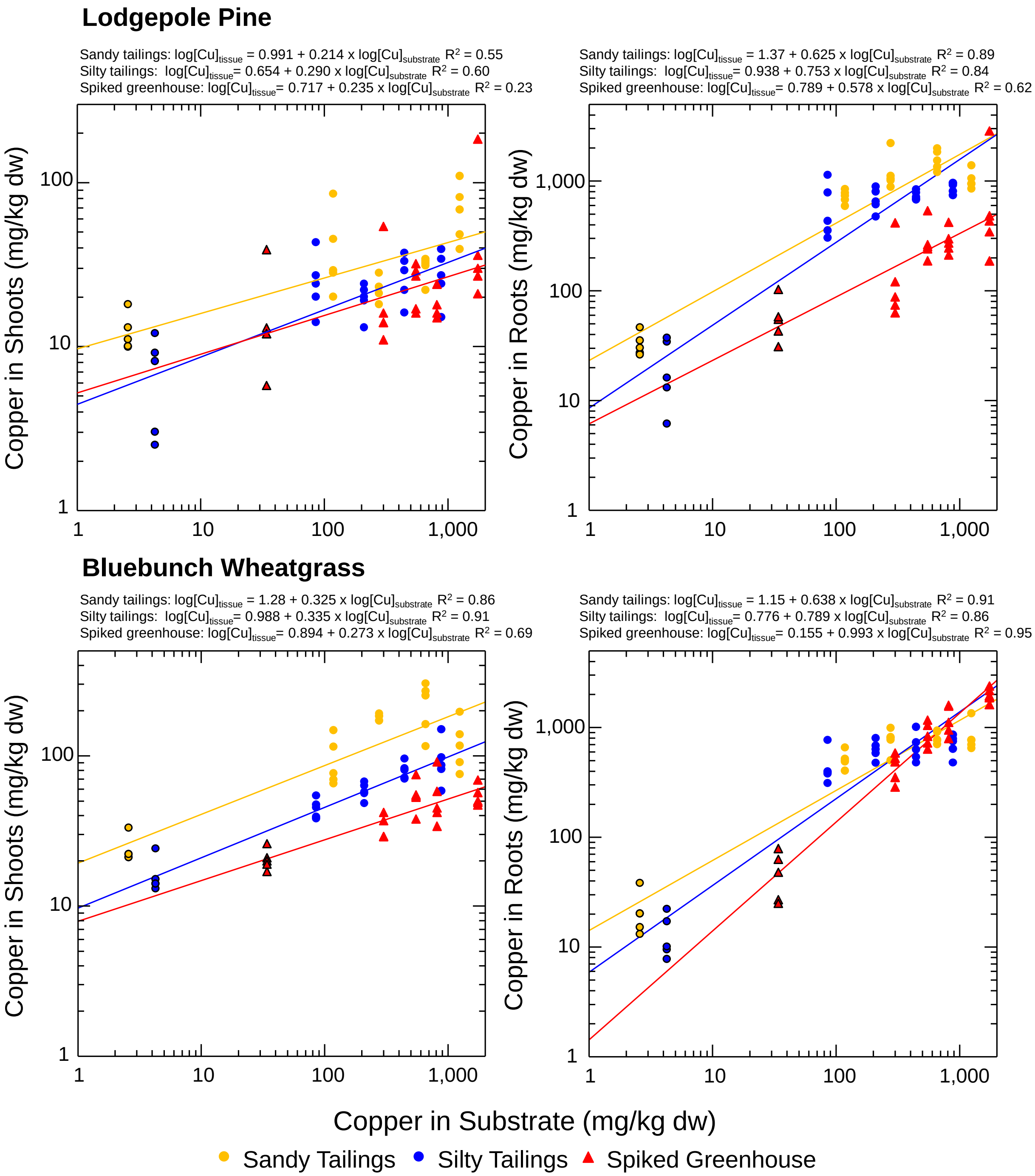

Figure 17: Relationship between Copper Concentrations Measured in Substrates and Plant Tissues during Toxicity Testing

Note: circles = diluted tailings, triangles = copper sulphate-spiked substrate, black outlined symbols = controls (same sand or silt controls for diluted tailings and spike substrate). Lines are best-fit log-linear regression for visual comparison and colours correspond to substrate type. Results are from individual relocates (*n*=5 for controls, tailings, and greenhouse soil). Measured concentration of copper in substrate for each treatment (*n*=1).

Control Sand (left), 10%, 25%, 50%, and 100% Sandy Tailings (right)

Control Silt (left), 10%, 25%, 50%, and 100% Silty Tailings (right)

Figure 18: Bluebunch Wheatgrass (Pseudoroegneria spicata) Growing in the Dilution Series of Sandy and Silty Tailings

The key findings are that:

- Copper associated with tailings has much less of an effect on plant growth than the same concentration added as a solution of copper sulphate. For example, the spiked concentration equivalent to that found in a 10% tailings sample (i.e., approximately 120 mg/kg Cu) caused high mortality to both lodgepole pine and bluebunch wheatgrass seedlings, and corresponding reductions in the growth endpoints in the surviving seedlings. The equivalent concentrations in tailings (diluted with silica sand) had relatively good performance that was not different than the negative controls.

- Although there were reductions in the growth endpoints relative to the negative controls in the 100% tailings samples, all plants ultimately germinated and demonstrated some degree of growth (Figure 18). An equivalent concentration of copper caused 100% mortality when spiked into the silty and sandy controls, but not when added to the greenhouse soil. This pattern highlights that organic carbon (in the form of the peat in the greenhouse soil) will bind copper and reduce its bioavailability.
- Overall, the relative patterns amongst the five different treatments show that copper appears to have a substantially lower bioavailability in tailings (and organic greenhouse soil) than the equivalent concentration in solution. This is consistent with the information about the minerology and geochemistry of tailings. The majority of toxicological data used to establish the numerical soil standard for protection of soil invertebrates and plants are based on spiked-soil toxicity tests or tests conducted using spiked solutions. These toxicity tests highlight the degree to which the numerical standard for copper (or any toxicity reference value that relies on spiked-soil data) can over-estimate the magnitude of effects associated with copper in tailings.

A final issue that is not evident in the figures relates to the physical stressors that are still present in the 100% tailings under field conditions, which were at least partially ameliorated by the methods used in the toxicity testing. The samples of tailings received by UBC required processing to allow homogenous mixing with the silica sands needed to create the dilution series. The sandy tailings were still moist throughout the sample container, but required the use of a trowel to break up the soil to allow for a homogenous mixing. The silty tailings were not consistently moist—there was a dried layer consisting of clumped soils. It was necessary to air-dry the entire sample and then reintegrate the resulting clumps into a homogenous sample using a rolling pin. UBC noted that the 100 and 50% tailings samples tended to form a surface crust during the 42-day exposure, and that the surface crust was more pronounced in the silty tailing samples. UBC also noted that the tailings samples tended to have higher moisture content in the bottom of the test vessels, with a corresponding tendency for the roots to seek out this layer and compete for water and nutrients. Surface drying was also observed in field investigations, but tended to form a ‘pan and crack’ layer where rainwater could still infiltrate via the cracks. Grasses were observed to grow preferentially in surface cracks in the field where growing conditions are more amenable.

This highlights the long-term benefits that will be expected to occur as organic carbon is reintroduced into the tailings. This reintroduction has been accelerated through the rehabilitation (e.g., mixing of tailings with the underlying original forest floor) but will also continue as vegetation is re-established. Humic colloid compounds are known to adsorb and complex with metals which mitigates toxicity of metals in organic soil (Besser et al. 2003). Soils with organic carbon are also able to maintain a more consistent moisture content and have a lower bulk density, which will reduce the physical effects noted above.

3.3.3.4 Risk Conclusion and Uncertainty Analysis

Golder concludes that the soil toxicity testing showed a more than a 20% reduction in growth is likely to occur in the tailings relative to the laboratory-based negative controls (e.g., pure silica sand or greenhouse soils). A field-collected reference soil (e.g., organic soil and/or mineral soils) was not included, and therefore, it is difficult to conclude whether the observed reduction in the laboratory would be ecologically relevant in the context of field conditions. The side-by-side trial demonstrated that copper is much less bioavailable in tailings than in a typical spiked-soil study, and that the plants appear to be able to maintain a consistent internal tissue concentration irrespective of the soil dose in tailings. Although the study was able to factor out the likely influence of nutrient depletion, other physical effects of the tailings (e.g., high bulk density leading to compaction; low organic carbon) are still likely exerting an influence on the toxicological endpoints.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Soil toxicity testing	Protect soil invertebrates and plants (collectively, the lower trophic levels) from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Measure emergence and growth in plant toxicity tests	Toxicological performance is not reduced by more than 20% relative to an appropriate grain size control.	Site wide	Lodgepole pine and wheatgrass survival and lodgepole pine shoot growth met decision criteria. Wheatgrass root and shoot growth, and pine root growth did not meet decision criteria.

There is a moderate level of uncertainty in extrapolating the magnitude of effects observed in the laboratory to the field. Toxicity testing and chemical analysis was conducted in a defensible manner consistent with standard practices. The deviations from the standard practices for a commercial chemistry or toxicology laboratory were relatively minor and were considered unlikely to impact the overall conclusions. The main uncertainty in this line of evidence relates to the fact that the study was ultimately based on two tailings samples collected from a single location which were tested with two representative species; however, tailings composition is reasonably homogeneous. The study design was reasonable and appropriate under the circumstances, but it was not designed to elucidate the relative influence of all possible factors that could influence plant survival and growth. However, even a multi-factorial design would not necessarily change risk management decisions, and ultimately, uncertainty in a given line of evidence is dependent on how the line of evidence was used in conjunction with the other information available.

There is a low degree of uncertainty with respect to the finding that copper bioavailability in tailings was limited. The use of the same concentration series with a variety of different diluted tailings and spiked soil treatments showed a consistent pattern between the tailings and spiked-soil relationships that was consistent with the literature as well as other lines of evidence.

3.3.4 Field Observations and Supporting Measurements

A field assessment was conducted in the summers of 2015 and 2016. The field assessment collected a variety of information about the overall health of the soil and plant community, including both quantitative and qualitative metrics that provide a baseline about the magnitude of effects associated with the initial deposition of tailings, as well as to help identify the relative influence of these physical factors versus chemical contaminants.

3.3.4.1 Study Design and Methodology

A field assessment was conducted by Golder scientists working in collaboration with Dr. Suzanne Simard, RPF. Dr. Simard is a Professor of Forest Ecology at UBC. The study design and methodology of the soil and plant community field assessments conducted in 2015 and 2016 are described in detail in Appendix H-1 and H-2, but in brief:

- A total of 21 study plots were established within the halo area. Plots were distributed across the two biogeoclimatic subzones present in the study area along transects established during soil sampling in 2014. A plot consisted of a circle with a radius of between 4 and 6 meters. The larger sized plots were established where the area contained trees in order to capture at least 15 trees within the plot area. A total of eight background plots were established. These were located at least 20 metres outside of the area of visual impact of the tailings material and were chosen to match the halo plot in terms biogeoclimatic site series, elevation, and slope.
- The plots were generally centered on a soil pit excavated for describing and sampling the tailings, forest floor and mineral soil. Field observations and measurements were collected according to established provincial protocols for terrestrial ecosystem descriptions (BC MFR & MoE 2010), and included:
  - In situ soil chemistry measurements (e.g., pH, conductivity, dissolved oxygen)
  - Forest floor and mineral soil properties (e.g., horizon presence and depths, rooting depths, soil mycorrhizae and fungi abundance and health)
  - Forest stand attributes (e.g., tree species composition, site index, height, age, basal area, stems per hectare, and stand structure)
  - Vegetation attributes (e.g., species cover, richness, and diversity).
  - Wildlife attributes and evidence of wildlife use (e.g., snag density and size, coarse woody debris cover and diameter, presence of tracks and scat, visual observations of wildlife).
- A statistical evaluation of the quantitative metrics was completed to determine if there were differences between the halo and the background locations (2015 data) or between years for the halo locations. Background plots were not re-examined in 2016. Nine of the 21 halo plots had received some degree of rehabilitation between 2015 and 2016. Rehabitation involved one or more activities such as removal of tailings, removal of dead standing timber, mounding or resloping, adding soil, adding coarse woody debris, and/or seeding. Rehabilitated plots were retained in the year to year comparison because the total number of plots was minimal[26]. Including rehabilitated plots is also consistent with the overall objective of tracking changes over time, which will include rehabilitation efforts.

3.3.4.2 Summary of Findings

A detailed summary of the methods, figures and representative photos from each plot, and a tabular summary of all field observations is provided in Appendices H-1 and H-2. A summary of selected metrics is provided below in Table 17.

[26] Four of the nine rehabilitated plots were logged to remove the dead standing timber, but were otherwise unaltered in terms of soils.

Table 17: Summary of Metrics from Soil and Plant Community Assessment

Metric	Area			Statistically Different	
	2015 Halo	2016 Halo	Reference	2015 vs 2016	2015 vs reference
Rooting Depth (cm)	16	23	30	No	Yes
Depth to Root Restricting Layer (cm)	6.4	70	0	Yes	Yes
Dissolved oxygen (mg/L)	5.2	6.0	8.5	No	Yes
pH	6.3–7.6	6.7–7.9	5.7	No	Yes
Conductivity (µs/cm)	190–323	40–106	39.6	Yes	Yes
Species richness	18	23	32	Yes	Yes
Shannon's diversity index	1.9	2.1	2.7	No	Yes

Broadly, the survey found:

- A distinct odour of sulphides and visual evidence of rotting material were present in the test pits dug in the halo in 2015. In situ measurements of dissolved oxygen and oxidation-reduction potential were lower than background, confirming a significant difference in the oxygen conditions in the root zone. A perched water table was present. The mechanism for these observations is the homogeneous, fine textured tailings were saturated with water and smothering the underlying forest floor, reducing oxygen concentrations in the soil, leading to the death of plant roots and soil organisms, and therefore, a switch to anaerobic metabolic pathways by facultative soil microbes. In 2016, the anaerobic odour was no longer present or had lessened. This corresponded with observations indicating drainage of the perched water table had occurred, that tailings were drier and the root restricting layer had, on average, receded below the native B horizon.
- Root decay was present in the halo in 2015. Average rooting depth was significantly shallower in the halo than the background plots, and roots were less abundant (Table 3 of Appendix H-1). The roots that were present in the halo had more saprophytic hyphae[27] than background. Ectomycorrhizae[28] were absent from the roots in most halo plots. Roots were not inspected in detail in 2016 for fungi, but a 44% increase in average rooting depth from 16 to 23 cm was noted relative to 2015. The presence of a growing root system is an important sign of recovery because it is inherently associated with a more robust soil foodweb community (including fungi, bacteria, and higher soil trophic levels) which would contribute to increased aeration, nutrient availability, metal sequestration, and an overall increase in productivity in the tailings/soil mixture.
- Abundance and richness of the shrubs and herbs community were significantly lower in the halo than background in 2015 with over forty plant species either eliminated or reduced by over 80% relative to background. Species gained in the halo between 2015 and 2016 tended to be "weedy" species, characteristic of disturbed areas (e.g., fireweed, dandelion, great mullein, hawksweed, white clover). Although there were more plant species identified in 2016 (87 versus 79 species) only 60 species were common to both years.

[27] Saprophytic hyphae refers to a network of growing fungi that process dead or decayed organic matter.

[28] Ectomycorrhizae are symbionts that form between fungi and roots of woody plants like pine, willow and birch. Colonization of roots by fungi to create ectomycorrhizae is an indicator of a healthy soil microbial community.

- There was further loss of berry-producing shrubs in the halo area. Cover values for these species were between 1 and 10% in 2015; all but one are now less than 1%. Seeds for these species would still be present in the original forest floor, and rehabilitation efforts to mix the soils and add organic material would be expected to bring seeds back to the surface where they can start to revegetate once the physical soil characteristics (e.g., moisture content, organic carbon, nutrients) are adequate. Saplings of many different shrubs are also part of the replanting program.
- There was an increase in the percent cover of mosses, lichens and livewort. There was roughly a 5-fold increase in the cover of this plant type. In particular, two mosses (knight's plume [*Ptilium crista-castrensis*] and hanging basket moss [*Rhytidiadelphus loreus*]), both important native species of surrounding forests, increased substantially from 2.7 to 11% and 0.75 to 22% cover. Growth of these mosses is a good signal of recovery of the plant community and indicates succession is proceeding well.
- Saplings of western red cedar (*Thuja plicata)*, hybrid spruce (*Picea engelmanii* X *glauca)*, and black cottonwood (*Populus balsamifera* ssp. *Trichocarpa)* were present in 48%, 57%, and 67% of halo plots in 2016, respectively. In addition to these three dominant species, saplings of subalpine fir (*Abies lasiocarpa)*, interior Douglas fir (*Pseudotsuga menziesii* var. *glauca)*, and paper birch (*Betula papyrifera)* were present in 5%, 10%, and 14% of halo plots in 2016, respectively. It is too early to confirm whether these saplings will establish themselves but their presence is an early indicator of the positive recovery trajectory of the forest community.
- Plant community development centred on old stumps and elevated logs, root wads and other organic material, and tended to be developed more rapidly in areas where tailings had been experimentally displaced (e.g., with a machine or at the soil pits dug with shovels in 2015). Plant community development was low or absent in areas with large amounts of coarse woody debris that had been placed following the TSF embankment breach. These areas have had insufficient time for the plant community to re-establish itself and recover.
- A heavy litter layer was observed overlying tailings deposits throughout the halo and especially in areas with dead standing trees. The soil substrate in halo areas where live trees were still present (i.e., there were more shallow deposits of tailings that did not reduce oxygen content) tended to have a darker, more “soil-like” colour. There were qualitative observations that the deposited leaf litter was being processed, and a greater amount of activity by earthworm and other invertebrates was noted.

3.3.4.3 Relative Contribution of Physical and Chemical Stressors

The deposition of tailings along the floodplan and halo areas of the Hazeltine channel resulted in the near-complete removal of vegetation in many areas, and immediate, significant impacts to the soil biological and vegetation communities along the halo. A key question in the risk assessment is to distinguish the influence of toxicological effects of metals from effects related to the physical nature of the deposited tailings. Toxicological effects, if any, may influence the long-term success of the revegetation process that is already underway. There were multiple observations that support the conclusion that the physical issues related to low oxygen and concomitant loss of the soil microbial community were more influential than toxicological effects related to copper (if any).

- The physical effect of tailings smothering the root zone was supported by the field measurements, as well as observations in the field in 2015 and 2016, which was consistent with the low oxygen mechanism:
  - Trees can change metabolic and biochemical processes to survive temporarily under conditions of low oxygen. Trees were observed to be alive and apparently healthy in the summer of 2014 after the TSF embankment breach, presumably as a result of adjusting their metabolism to low oxygen soil conditions and as a result of entering winter dormancy early.
  - Trees in the halo areas with the deepest tailings deposits died in early-spring of 2015. Snow melt and spring rains would likely increased the water content of the tailings deposits, thus creating fully saturated conditions (i.e., a perched water table) and increasing oxygen depletion. The shift from hypoxic to anoxic soil conditions would have amplified root mortality which meant that trees would have been unable to acquire sufficient water to meet demands for photosynthesis and transpiration in spring.
  - As trees flushed (i.e., annual spring growth started) and soils warmed, the foliage would have continued to transpire even though roots were incapable of acquiring and transporting sufficient water to the crown. Without sufficient water conductance, the entire crown would die rapidly, leading to tree mortality over a period of one to two months. This pattern was observed in all tree species.
- If copper toxicity was the dominant mechanism, plants in the field would be expected to show foliar interveinal chlorosis (i.e., a yellowing of otherwise healthy appearing leaves because of the lack of chlorophyll). The relationship between chlorosis and copper accumulation has been demonstrated in many plants (Eleftheriou and Karataglis 1989; Panou-Filotheou et al. 2001), including oak (Wiseniewski and Dickinson 2003), maple, dogwood and pine (Heale and Ormrod 1982). Chlorosis was not observed during any of the field investigations. The rapid death of many different species of trees from areas with differing copper concentrations does not fit the pattern of copper toxicity effects, which would have involved some degree of chlorosis, in combination with stunting of new growth. Mortality would be been expected to occur over a longer time frame.

3.3.4.4 Risk Conclusion and Uncertainty Analysis

There were multiple signs of recovery in the survey. Indicators included increased cover of mosses and lichens, as well as germination of tree saplings. Another indicator was that the presence of nitrogen-fixing alder (particularly important to improving plant available nutrients) was noted for the first time in 2016, however, the abundance was still low (<1%). There has also been recolonization by weedy species. An ecosystem-level, quantitative metric was used by Golder in cooperation with Dr. Simard. Assessment of root health involved a variety of quantitative measurements (e.g., dissolved oxygen, rooting depth) with semi-quantitative indicators such as abundance of ectomycorrhizae. There are no regulatory standards for plant health measurements that represent an "acceptable" level of impact. Interpretation of these types of data involves professional judgement by the subject matter expert. It was not practical to try and establish a quantitative relationship between percent tailings and "acceptable productivity of forest ecosystem" beyond testing whether or not soils in the halo zone meet the established inference in forest ecology that a healthy soil microbial community will result in a healthy soil invertebrate and plant community.

There were multiple indicators noted in the survey in 2015 that indicated adverse effects were present and all data collected supported the hypothesis that the primary mechanism of action was related to a physical impact related to the deposition of tailings. Copper was considered unlikely to have been a significant factor relative to these physical impacts. Rehabilitation efforts are focused on mitigating this physical mechanism of action, and there were indicators during the 2016 survey that root health was improving. Golder has a high degree of certainty in the overall conclusion that recovery has started, but acknowledges that the magnitude of impact related to physical stressors is still substantive and that it may take a considerable length of time for the impacted areas to return to an acceptable level of ecological function.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Field observations	Protect soil invertebrates and plants (collectively, the lower trophic levels) from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Evidence of chlorosis in plants exposed to tailings	Qualitative	Site wide	No visual observations that chlorosis was present in wild plants or in soil toxicity testing
		In situ measurement of forest health indicators	Qualitative	Site wide	Tailings deposition resulted in reduced health of forest ecosystem. Indicators of recovery are present (e.g., deeper rooting depth observed in 2016)

3.4 Effects-Based LOE: Higher Trophic Level Receptors

3.4.1 Amphibian Salvage

The PEEIAR included an evaluation of the habitat types, vegetation and wildlife species within the impacted areas (SNC Lavalin 2015, included in MPMC 2015). SNC Lavalin (2015) used terrain ecosystem mapping to document the ecosystem types within the impact areas. These ecosystem types were then described in terms of the presence of rare or sensitive species or ecological assemblages, either based on site-specific data, or the best available information about the types of species that would be affiliated with each ecosystem type. SNC Lavalin (2015) concluded that there was 4.9 hectares of wetland present in the impact area prior to the TSF embankment failure. This wetland consisted of a scrub birch/sedge/peat moss subunit of the Interior Cedar Hemlock (ICHmk3-BS), described as a nutrient poor organic wetland in depressions associated with a near-surface water table. It is dominated by sphagnum mosses, scrub birch and sedges, associated with Labrador tea, march cinquefoil and bog cranberry. The location of this former wetland was in the Polley Plug (3 hectares in total) and upper Hazeltine area (1.9 hectares in total). MPMC (2015) also included an evaluation of the aquatic habitat associated with Edney and Hazeltine Creeks that was impacted. The total area of riparian habitat altered by the TSF embankment breach was estimated at approximately 2 hectares for Edney Creek and 72 hectares for Hazeltine Creek. SNC Lavalin (2015) detected common gartnersnake, Pacific treefrog, western toad, and Columbia spotted frog during the field surveys after the TSF embankment breach. Tailed frogs and painted turtles are also possible herpetofauna in the ICHmk3 biogeoclimatic zone, and in total, this biogeoclimatic zone has seven possible amphibians and six possible reptile species (BC Ministry of Forests, 2017).

Numerous tadpoles were observed in ponds or pools of water in 2015 that formed in depressions throughout the Hazeltine corridor, particularly in the Polley Flats. A permit to live-capture, temporarily possess, transport and release amphibians was obtained in 2016 prior to planned rehabilitation and re-vegetation of the Polley Flats in the upper portion of Hazeltine corridor in 2016. The tadpoles were confirmed to be western toads and were the dominant species. It was therefore necessary to capture amphibians from inside the Hazeltine corridor and relocate them to nearby undisturbed habitat to reduce potential for harm and mortality due to rehabilitation work.

Results of the amphibian salvage surveys are provided in Table 18. A total of 40 standing water depressions with amphibians were found (Appendix G-2). Three amphibian species, including western toad, were observed in ponds within Polley Flats (upper Hazeltine corridor). Western toad is provincially blue-listed and federally designated as Special Concern under Schedule 1 of the *Species at Risk Act* (*SARA*; BC Conservation Data Center 2016). Western toad can be found in a wide diversity of habitat types, and use stream margins, wetlands, backchannels, ponds, and human-made infrastructure (e.g., ditches and road ruts) to lay their eggs. Eggs metamorphize in 4 – 12 weeks; the adults can then migrate several kilometers to find foraging areas. Adults hibernate underground and then congregate back in their breeding sites. SNC Lavalin (2015) concluded that the presence of western toads during the post-impact surveys indicated that they could recolonize the study area if habitat was present. Observations during the subsequent rehabilitation and the requirement to conduct a significant western toad salvage program supports a conclusion that western toad can rapidly occupy the available habitat, even if the habitat is relatively poor in complexity or quality. The natural ecology of the corridor prior to the TSF embankment breach did not contain large amounts of amphibian habitats, and therefore, the standing water depressions were recontoured to remove this attractant habitat.

Table 18: Summary of Amphibians Salvaged from 15 June to 16 August 2016

Amphibian Species	Number of Amphibians Salvaged by Life-Stage				
	Tadpoles	Juveniles	Sub-adults	Adults (non-breeding)	Total
Western Toad *(Anaxyrus boreas)*	65,997	8,893	323	54	75,267
Long-toed salamander *(Ambystoma macrodactyllum)*	284	30	2	-	316
Columbia spotted frog *(Rana luteiventris)*	58	-	15	16	89

The presence of large numbers of amphibians in two successive years provides a qualitative indicator that amphibians are able to successfully survive, grow and reproduce despite direct contact with tailings and surface water in contact with tailings. The data above were collected as part of amphibian salvage efforts during remediation work in Polley Flats and therefore was not a study that was designed to measure growth, reproductive success, or survival in different ponds were made. No assessment of the quality of habitat and its ability to sustain a western toad population over the long term has been made. A detailed survey to determine the presence or absence of other herptofauna in the impacted area has not been conducted. Exclusion fencing has been erected along portions of the Hazeltine corridor, particularly in the Polley Flats area to limit further access by toads so that remedial excavating can continue without harming amphibians that seem to be attracted to these areas.

3.4.2 Food Chain Modelling

3.4.2.1 General Approach

A mechanistic food chain model was used to estimate the potential hazard associated with the ingestion of copper and vanadium by wildlife as a result of the consumption of food, water and soil. A detailed discussion of the food chain model and its parameterization is provided in Appendix J, but in brief:

- A number of different wildlife species were selected to represent different parts of the ecosystem, as described in the problem formulation (Section 2.3.3.4).
- The daily ingested total dose was estimated with mathematical equations for each receptor based on standardized ingestion rates and dietary preferences.
- Site-specific information about the concentrations of COPCs in various dietary items (e.g., plants, soil invertebrates) was used to the extent possible. If site-specific data were not available, uptake models from the literature or derived from site-specific data were used.
- A "safe" daily ingested dose (i.e., a toxicity reference value) was derived from the toxicological literature. A toxicity reference value that reflected "no observed adverse effects" was applied to rare, threatened or endangered species to provide a level of protection for individual organisms. A "lowest observed adverse effect"-based toxicity reference value was applied to common species that are protected at the population level. The magnitude of effects associated with a "lowest observed adverse effect level" is often limited and not necessarily indicative of an actual adverse effect outside the laboratory.
- Hazards for each wildlife receptors were estimated using a hazard quotient approach. A hazard quotient of less than 1 means risks to wildlife associated with the ingested dose would be unlikely to be present. A hazard quotient that is substantially greater than 1 means that the ingested dose may be hazardous to wildlife, and that further refinement of the food chain model, or further evaluation of risks to wildlife with other lines of evidence may be warranted. A hazard quotient that is marginally greater than 1 does not mean that risks to wildlife are present—the risk management decision should consider the magnitude of the hazard quotient in light of the degree of conservatism present in the food chain model parameterization. Hazard quotients were also calculated for background conditions to provide additional context.

3.4.2.2 Summary of Food Chain Model Parameterization

Detailed information regarding the food chain parameterization and model inputs is provided in Appendix J. In brief:

- Receptor variables such as body weight, ingestion rates, and dietary preferences were adopted from Canadian risk assessment guidance (ECCC 2012c) wherever possible. Reputable sources were used if a default value from ECCC (2012c) was not available.
- Realistic worst-case exposure concentrations were calculated for soil, water and the various dietary items. These were typically based on the 95% UCLM concentration. Concentrations in plants and soil invertebrates in the rehabilitated areas were estimated using the 95% UCLM soil concentration and site-specific uptake models or bioaccumulation factors. Concentrations in small mammals (a dietary item for carnivores) were estimated using an uptake model from the literature.

- Olive-sided flycatcher and common nighthawk were added as receptors in the food chain model at the request of the Ministry. Both species are listed, but were not originally included because their diet consists almost exclusively of aerial insects. There was no site-specific tissue data for aerial insects available, and no bioaccumulation factors for aerial insects were available from the literature, and therefore, it was necessary to make the assumption that aerial insectivores were consuming litter-dwelling invertebrates.
- Habitat range was ultimately not incorporated in the food chain model because the total area of the impact was larger than the default habitat range for all receptors.
- The relative bioavailability of water, soil and dietary items was incorporated. Water was assumed to have a relative bioavailability of 100%. Soil was assumed to have a relative bioavailability that matched the average bioavailability as measured in site-specific PBET tests. Dietary items were assumed to have a relative bioavailability of 40% for copper and 10% for vanadium based on consideration of the available literature.
- Toxicity reference values were derived from the ecological soil screening level database. The ecological soil screening level database is a recommended data source for ecological risk assessment per Technical Guidance 7.

Table 19: Wildlife Receptors Retained for Food Chain Modelling

Receptor Group	Feeding Guild	Surrogate Receptor
Birds	Herbivorous	Dark-eyed junco
	Insectivorous	Robin, olive-sided flycatcher, common nighthawk
	Omnivorous	Common raven
	Carnivorous	Northern pygmy-owl
Mammals	Herbivorous	Meadow vole, Mule deer
	Insectivorous	Masked shrew
	Omnivorous	Deer Mouse
	Carnivorous	Ermine, Fisher[a]

(a) Listed species.

Piscivorous wildlife were also considered using a food chain model and are described in the aquatic risk assessment (Section 4.8.1).

3.4.2.3 Inclusion of Post-Rehabilitation Soil Exposure Conditions

The model incorporated the influence of ongoing terrestrial rehabilitation. Rehabilitation to date has involved a variety of techniques, including transplantation of soils, intermixing of the tailings with the underlying native soils, and scarification with some integration of natural soils into the areas along the slopes of the corridor. All three methods result in lower bulk soil concentrations of metals, which would result in lower concentrations in plants and soil invertebrates. In brief:

- Soil concentrations for the untreated area were characterized by samples collected between 2014 and 2016 (Appendices A, E-1, and E-2). The untreated area was subdivided in three zones (Polley Flats, upper Hazeltine, lower Hazeltine) to improve the spatial resolution. Soil concentrations from the three rehabilitation treatment zones (transplanted, intermixed, scarified) were characterized by samples collected in November 2016 (Appendix G-1).

- The food chain model was run separately for each area (untreated Polley Flats, untreated upper Hazeltine, untreated lower Hazeltine, transplanted, intermixed and scarified) and a hazard quotient was calculated for each treatment.
- An overall hazard quotient that applies to the population as a whole across the entire study area was calculated based on area weighting (untreated Polley Flats, 50 ha; untreated upper Hazeltine, 35 ha; untreated lower Hazeltine, 40 ha; transplanted soils, 9 ha; intermixed soils, 49 ha; and scarified soils, 33 ha). An assumption of this calculation is that terrestrial wildlife receptors were assumed to access and spend time (proportional to the area) in all six areas.

3.4.2.4 Summary of Findings

Hazard quotients are summarized in Table 20 for copper and Table 21 for vanadium. These tables provide the overall area-weighted hazard quotient, as well as the hazard quotients for the subsidiary areas. The hazard quotients based on the available background concentrations are also provided for context.

Table 20: Summary of Hazard Quotients for Terrestrial Wildlife – Copper

Receptor	Species	Untreated Polley Flats	Upper Untreated Corridor	Lower Untreated Corridor	Transplant Soils Zone	Intermixed Soil Zone	Scarified Zone	Area-weighted Hazard Quotient	Background Forest
Percent of Hazeltine Corridor (as of Nov 2016)		23%	16%	19%	4%	23%	15%		
Birds	Dark-eyed junco	0.92	0.85	0.47	0.17	0.28	0.20	0.54	0.19
	American robin	**2.9**	**2.5**	**1.8**	0.46	0.85	0.56	**1.7**	0.60
	Common nighthawk*	**3.3**	**3.3**	**1.3**	0.47	0.60	0.50	**1.8**	0.90
	Olive-sided flycatcher*	**4.5**	**4.5**	**1.8**	0.64	0.83	0.69	**2.4**	**1.2**
	Common raven	0.30	0.27	0.15	0.06	0.09	0.07	0.17	0.08
	Northern pygmy-owl	0.16	0.16	0.15	0.13	0.15	0.13	0.15	0.11
Mammals	Meadow vole	0.60	0.53	0.32	0.12	0.2	0.14	0.36	0.12
	Masked shrew	**5.1**	**5.0**	**2.3**	0.75	**1.1**	0.85	**2.8**	**1.3**
	Deer mouse	0.91	0.85	0.42	0.14	0.22	0.16	0.50	0.22
	Ermine	0.13	0.12	0.12	0.10	0.12	0.10	0.12	0.09
	Fisher*	0.08	0.08	0.08	0.06	0.07	0.06	0.08	0.06
	Mule deer	0.06	0.06	0.05	0.02	0.03	0.02	0.04	0.02

Bold and underlined values: indicate a hazard quotient greater than 1.

* Listed species.

Table 21: Summary of Hazard Quotients for Terrestrial Wildlife – Vanadium

Receptor	Species	Untreated Polley Flats	Upper Untreated Corridor	Lower Untreated Corridor	Transplant Soils Zone	Intermixed Soil Zone	Scarified Zone	Area-weighted Hazard Quotient	Background Forest
Percent of Hazeltine Corridor (as of Nov 2016)		23%	16%	19%	4%	23%	15%		
Birds	Dark-eyed junco	0.26	0.25	0.12	0.09	0.05	0.05	0.15	0.05
	American robin	0.91	0.84	0.77	0.29	0.20	0.18	0.57	0.17
	Common nighthawk*	0.41	0.41	0.27	0.032	0.054	0.03	0.23	0.019
	Olive-sided flycatcher*	0.56	0.56	0.37	0.04	0.07	0.05	0.32	0.03
	Common raven	0.06	0.06	0.04	0.03	0.02	0.02	0.04	0.02
	Northern pygmy-owl	0.07	0.07	0.06	0.04	0.05	0.04	0.06	<0.01
Mammals	Meadow vole	0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Masked shrew	0.08	0.08	0.06	0.02	0.01	0.01	0.05	<0.01
	Deer mouse	0.02	0.02	0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Ermine	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Fisher*	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Mule deer	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01

Bold and underlined values: indicate a hazard quotient greater than 1.

*Listed species.

3.4.2.5 Wildlife Observations

Food chain models are improved when there is confirmation that the selected receptors are in fact representative of wildlife species that occupy and utilize the site. Wildlife were observed during the soil and plant community assessments (Appendix H-2). Within each plot signs of wildlife were recorded, such as direct sightings, auditory signs, tracks, droppings, burrows or dens. The frequency of observations was calculated for comparison between the 2015 and 2016 programs to determine if use by wildlife had increased or decreased. The frequency of deer, small mammals, and birds tended to increase from 2015 to 2016, which was attributed to increased forage availability. Site staff have also observed wildlife using the areas.

3.4.2.6 Uncertainty Analysis

Food chain modelling was the primary line of evidence used to determine if COPCs were likely to bioaccumulate in the food chain to the point that they present an unacceptable hazard to wildlife receptors. Food chain models are effectively a measure of exposure where the cumulative dose (from food, soil and water) is compared to a "safe" dose taken from the literature. The food chain model, despite its complexity, is equivalent to comparing a measured soil concentration to a conservative guideline value. This is a challenge because the evaluation of risks to wildlife receptors lack laboratory- or field-based measures of effect to place the calculated hazard in context. This lack of context results in the hazard quotient being used as the sole basis of risk characterization, which means:

- **There is a tendency to ignore the compounding effect of making many individual, conservative decisions on the overall uncertainty of the hazard quotient.** The level of conservatism in the parameterization of a food chain model would generally be higher in a screening-level risk assessment than in a detailed risk assessment because the purpose of the screening-level risk assessment is to be able to confidently eliminate receptors/contaminant combinations from further evaluation. However, this does not mean that all parameters ought to be set to upper-bound values, because the resulting hazard quotients would become inflated by the product of each conservative decision (i.e., the conservatism of the hazard quotient would be more than the sum of the conservatism of each variable). It is important to balance the decision-making in the parameterization such that the total uncertainty in the resulting hazard quotients provides adequate confidence that risk management decisions are supported by the available science without inadvertently generating excessively high hazard quotients that are toxicologically implausible. There are few examples in the literature where hazard quotients from a food chain model have been "validated" by direct measurement of effects, and where such examples exist, the balance of evidence is that food chain models will tend to over predict the actual toxicological hazards (Tannenbaum 2003).
- There is a tendency to interpret hazard quotients as a binary outcome where any hazard quotient less than 1 is considered to be "acceptable risk" and any hazard quotient greater than 1 is considered to be "unacceptable risk". This inference of risk based on a hazard quotient may be a common practice for evaluating some lines of evidence as part of a screening-level risk assessment, but is particularly problematic in food chain models. A hazard quotient that is marginally greater than 1 does not mean that risks are present, particularly in light of the degree of compounding conservatism in the food chain model.

A detailed discussion of how the uncertainty in each variable in the food chain model has contributed to the overall degree of conservatism is provided in Appendix J. Selected variables with noteworthy uncertainty are summarized below:

- **Dietary preferences and availability of site-specific tissue concentrations.** Each wildlife species is assumed to consume a variety of different dietary items based on information from the literature. The exact proportion of dietary items consumed by wildlife in the study area is unknown, and likely more complex than the assumptions in the food chain model because wildlife species do not seek out a balance of dietary items (they would tend to opportunistically feed on what is available within broad preferences). Tissue chemistry data was available for several different dietary items which are assumed to be a representative surrogate for those broad preferences. There were several instances where it was necessary to use highly conservative assumptions—in particular, risks to aerial insectivorous birds (flycatcher, nighthawk) assumed that dietary concentrations would reflect litter-dwelling invertebrates. This assumption was necessary because there was no site-specific data for aerial insect tissue concentrations available.

- **Relative bioavailability.** The ingested dose of copper and vanadium was adjusted to consider the relative bioavailability. Relative bioavailability, in this application, refers to the assumption that the original toxicological study from the literature has a 100% bioavailable ingested dose. The original toxicological study typically involves spiking a known amount of a water-soluble copper salt into food (e.g., copper sulphate spiked into commercial rabbit food). The assumption is that 100% of that spiked copper is absorbed by the test animal for the purposes of calculating the toxicity reference value. The relative bioavailability of copper concentrations in soil, plants and invertebrates is expected to be substantially less than copper sulphate spiked into commercial rabbit food. The food chain model assumed that water would have a relative bioavailability of 100%, that dietary items would have a bioavailability of 40% based on the available literature, and that soil would have a relative bioavailability between 36 and 44% based on site-specific physiological based extraction tests (PBET). The average of values from the literature or site-specific PBET testing were used in the food chain model so that this variable would not unnecessarily over- or under-estimate the calculated risks.
- **Selection of toxicity reference value.** There are multiple approaches for deriving a toxicity reference value which vary in their degree of conservatism and their ability to incorporate the totality of the available mammalian or avian toxicity data from the literature. No specific requirement for a derivation is specified by regulation and this is an area of considerable professional judgement. The toxicity reference values reflect the lowest available toxicity data points summarized by ECO-SSL that correspond to a lowest observed adverse effect level (LOAEL) for common receptors, and a no observed adverse effect level (NOAEL) for rare or endangered receptors that warrant protection at the individual organism level. There are multiple derivation approaches available that would use all available toxicological data (e.g., a species sensitivity distribution) which would result in equally defensible (but potentially higher) toxicity reference values. The current approach is intentionally conservative, consistent with the expectations for a screening risk assessment.

Overall, Golder concludes that the food chain model was parameterized in a manner that provides an appropriate balance and uses the available site-specific data to the extent possible. Changes in any of the parameterization could result in an increase in the calculated hazard quotient (e.g., concentrations or bioavailability values increase) or a decrease in the calculated hazard quotient (e.g., concentrations or bioavailability values decrease or toxicity reference values increase). The current hazard quotients represent a realistic estimate of potential risks based on the available data, and as noted above, it would be inappropriate to set all variables to a conservative, upper-bound values, because the resulting hazard quotients would become inflated by the product of each conservative decision. The food chain model has balanced the parameterization such that the total uncertainty in the resulting hazard quotients provides adequate confidence that risk management decisions are supported by the available science without inadvertently generating excessively high hazard quotients that are toxicologically implausible.

3.4.2.7 Risk Conclusion

Golder concludes that risks to wildlife receptors associated with ingestion of soil, water and dietary items are low based on the calculated hazard quotients. The food chain model was constructed with reasonable worst-case exposure concentrations and appropriately conservative assumptions. Hazard quotients were less than one for all receptor/contaminant of potential concern combinations with the exception of:

- Copper/robin had hazard quotients between 1.8 and 2.9 for the three untreated areas and an aggregate hazard quotient of 1.7. This is considered to be a low risk.
- Copper/shrew had hazard quotients between 1.1 and 5.1 for the three untreated areas and the intermixed mixed area, and an aggregate hazard quotient of 2.8. This is considered to be a low risk to the shrew population, especially considering that the background area had a hazard quotient of 1.3.
- Copper/flycatcher had hazard quotients between 1.8 and 4.5 for the three untreated areas and an aggregate hazard quotient of 2.4. This is considered to be a low risk to individual flycatchers, in light of the fact that the background area had a hazard quotient of 1.2, and the model assumed that this aerial insectivore is consuming litter-dwelling invertebrates.
- Copper/nighthawk had a hazard quotient between 1.3 and 3.3 for the three untreated areas, and an aggregate hazard quotient of 1.8. This is considered to represent a low risk, in light of the conservative assumption that this aerial insectivore is consuming litter-dwelling invertebrates.

Remediation of these untreated areas is ongoing, and is expected to result in a lower aggregrate hazard quotient over time as the remediation and restoration continues.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Food chain model	Protect terrestrial wildlife from reduction in survival, growth and reproduction as a result of exposure via ingestion of food items, incidental soil and drinking water.	Calculate daily ingested dose for variety of mammal and bird receptors.	Daily ingested dose is less than toxicity reference value (HQ < 1) on a site-wide basis	Site wide	Risks to majority of wildlife receptors is negligible (i.e., HQ is less than 1). Risks for four receptors from copper is low based on calculated hazard quotient.

3.5 Risk Conclusions

3.5.1 Weight of Evidence Process

The following sections provide a summary of the weight and magnitude of response for each line of evidence as applied to each major ecosystem component. The weight of evidence approach is based on guiding principles described by provincial guidance supporting Protocol 20 (SABCS 2008, 2011) and was described by Golder during the engagement with technical reviewers (Appendix F-2). In brief, the weight of evidence process started with assigning a rating of “high”, “moderate” or “low” to each line of evidence based on weight, magnitude and overall uncertainty.

- A preliminary weight was assigned in the problem formulation by considering: a) how closely the line of evidence matched the assessment endpoint, and b) the likely uncertainty associated with the line of evidence based on sensitivity and specificity, study design and data quality objectives, representativeness, and correlation and causation. These preliminary weights were re-evaluated at the conclusion of the exposure and effects assessment based on the overall quantity and quality of information obtained.
- The magnitude of response for each line of evidence was evaluated by comparing the data to a decision criterion that specified when the magnitude of the observed effect would be considered a potential risk. Some decision criteria are specified by provincial guidance, while others required consideration of the available literature to determine a criterion that was ecologically relevant. Decision criteria can be quantitative or qualitative, depending on the circumstances of each line of evidence.
- The uncertainty in the line of evidence was based on the degree to which it was used to inform the risk assessment conclusion. Broadly, a line of evidence was considered more certain if the data were sufficient to extrapolate from specific sampling locations to a broader spatial scale, if it was able to clarify the relative influence of physical versus contaminant stressors, and if it there was a quantitative basis for evaluating the magnitude of response against a regulatory-approved decision criterion.

Weight-of-evidence assessment does not need to be a quantitative, prescriptive process. Best professional judgement is part of the process. To provide transparency, each individual line of evidence was accompanied by a narrative that describes the site-specific data and relevant information from the peer-reviewed literature. A key question in the narrative is the degree to which the available data support a causal relationship. Most decision frameworks incorporate Hill's (1965) criteria for causation in one form or another. Environment Canada (2013b) provide guidance on integrating causality into ecological risk assessments for contaminated sites. Establishing causality is itself a weight of evidence based on observational data, experimental results from manipulation, or alignment with general knowledge from the scientific literature (Environment Canada 2013b). Forbes and Calow (2004) conclude that causality is present if there is a correlation in field data that can be confirmed through a controlled experiment and that is consistent with the known scientific information. If the evidence for causality is weak, it would be difficult to establish a meaningful remedial strategy where actions related to a specific contaminant or physical stressor would be likely to result in biological improvement.

3.5.2 Consideration of Physical Stressors and Contaminant Bioavailability in the Risk Assessment

Scouring and deposition of tailings resulted in significant physical impacts to all components of the soil ecosystem (microbes, fungi, invertebrates and plants) within the spatial bounds of the affected area. Golder concluded that is not necessary to quantify the magnitude of this impact (beyond the qualitative observation that it was substantive and wide-spread) as part of this ecological risk assessment. Information about the specific areas of impacted habitat are described in MPMC (2015).

The purpose of the risk assessment was to determine if the contaminant stressor (i.e., residual concentrations of metals in tailings) presented an unacceptable risk that requires risk management over and above the rehabilitation and monitored recovery program that is already underway to address the physical stressors (nutrient and physical deficiencies) that are inherent from the scouring and deposition event. Broadly, a terrestrial ecosystem will gradually recolonize an impacted area (i.e., the area impacted by tailings, or areas impacted by natural phenomenon like landslides) over multiple waves of colonization. An in-depth review of ecological theory on reclamation of mine tailings was outside the scope of this risk assessment, but a simplified overview is that early waves focus on plants and microbes that can tolerate the physical and nutrient deficiencies and help to rebuild the organic carbon and soil structure; then, a second wave focuses on a more diverse soil community that includes invertebrates and fungi that process the organic carbon and rebuild nutrient cycling; then, a third wave involves re-establishment of more a more diverse soil invertebrate and plant community that includes predators and herbivores; and finally, the soil community starts to achieve a more natural ecological function that continues to develop over the long-term.

An unacceptable risk, in this context, was related to whether the effects from chemical contamination or physical stressors will prevent the long-term successful re-introduction of a biologically diverse, functional, self-sustaining and inter-dependent soil community. The bioavailability of copper is a key consideration in the risk assessment. There were multiple lines of evidence in the ecological risk assessment that showed that copper has relatively low bioavailability. In summary, the main lines of evidence included:

- Geochemical modelling and leach testing found that copper associated with tailings was not likely to dissociate into soil porewaters. Water soluble fraction data confirmed the findings from the geochemical model that the amount of copper that would dissolve into the porewater was relatively lower
- Physiologically based extraction tests showed that less than 100% of the copper ingested by a bird or mammal would become available for uptake from the gastrointestinal tract
- The bioaccumulation factors derived from site-specific collection of soil and plant samples were all less than 1, and inversely correlated with soil concentrations (i.e., more copper in soil does not mean more copper will be accumulated by plants).
- Laboratory-based toxicity tests showed that copper accumulation by plants from tailings was low. The experiment included side-by-side testing of tailings and copper spiked into a similar grain size. The bioavailability of copper from tailings was far less than the same concentrations added as a solution.

3.5.3 Terrestrial Plants

The majority of the lines of evidence in the overall risk assessment focused on terrestrial plants because they are expected to be a primary part of the initial stages of ecosystem recovery. The weight of evidence conclusion was that the presence of residual metals in soil are not likely to prevent reintroduction of a biologically diverse, functional, self-sustaining and inter-dependent plant community into the areas impacted by the physical stressors. The physical and nutrient deficiencies of the tailings are expected to influence the re-establishment of a functioning soil ecosystem more than residual metal contamination. Rehabilitation to improve these physical and structural soil deficiencies is underway, and there are positive indicators that natural and facilitated re-establishment of vegetation is occurring.

A summary of each individual line of evidence that was considered in reaching this conclusion is provided in Table 22 and the following information provides a narrative overview of how these various lines of evidence were integrated into the overall conclusion. A summary of each individual line of evidence that was considered in reaching this conclusion is provided in Table 22 and the following information provides a narrative overview of how these various lines of evidence were integrated into the overall conclusion. This evaluation focused on determining the degree to which metals or physical stressors demonstrated causality—was there a correlation in field data that can be confirmed through a controlled experiment and that is consistent with the known scientific information? Key points that were considered:

- There were multiple lines of evidence that showed metals associated with tailings had limited bioavailability. There was coherence between the mechanistic theory of how the tailings would likely behave in the environment (the geochemical conceptual model) which was confirmed with experimental evidence (leach and other testing). There were no indications in the geochemical assessment that metals would mobilize at an appreciable rate out of tailings over time, and every indication that ongoing remediation efforts would further reduce metal mobilization.
- Bioaccumulation of metals is occurring in plants, but at a relatively low magnitude because of low bioavailability. There was a predictable relationship that could be used to predict the amount of copper in plant tissues based on soil concentrations (bioaccumulation model). Copper bioaccumulation by plants has been studied at many contaminated sites and there is a standardized model based on the literature that is frequently used for risk assessments in British Columbia. The standardized model from the literature was not needed because site-specific data were available. The site-specific relationship was less steep and tended to plateau at higher concentrations than the standard model. In other words, copper is accumulating in plants at lower concentrations than predicted by the model and increasing the copper concentrations in soil does not consistently result in a commensurate increase in plant concentrations. This plateau was observed in the field collected samples as well as in data collected during the soil toxicity testing. Vanadium was less bioavailable than copper—there was no relationship between soil and tissue concentrations.
- Concentrations of copper and vanadium in soil exceeded numerical standards for the protection of soil invertebrates and plants, which are based on toxicity test data. Soil toxicity testing was conducted in the laboratory using representative species (grass and pine) to test if the magnitude of effects was consistent with the literature used to set the numerical standard. The growth of plants was lower in site samples than in an artificial control, but both species were able to germinate successfully and were able to grow to some degree. The toxicity testing program included a side-by-side trial which demonstrated that the bioavailability of copper in tailings was much less than the toxicological studies used to derive the numerical standard. The toxicity testing program also supported the hypothesis that observed effects on plants (in the laboratory or in the field) can be at least partially attributed to the physical deficiencies of the substrates. Site-specific samples had a bulk density that was near the literature-based threshold value for when bulk density can present a barrier to root growth. Finally, a case study from the literature showed inert tailings (in the absence of metal contamination) similar to the site had long term effects on plant growth because of nutrient and physical deficiencies.

Table 22: Summary of Lines of Evidence for Terrestrial Plants

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there bulk concentrations of metals that are hazardous in soils?	Quantitative – Calculate hazard quotient by comparing soil chemistry to CSR numerical standards for protection of soil invertebrates and plants (or generic standards).	Low. A numerical standard for total metal based on toxicological literature does not account for site-specific bioavailability. The exceedance of a numerical standard does not mean that adverse effects to organisms is actually occurring. The linkage to the assessment endpoint is extremely low for vanadium because it is a generic standard, not a pathway-specific standard.	High for copper. The maximum soil concentrations are up to 10x the standard, and the 95% upper confidence of the mean was approximately 5x the standard. Low for vanadium. The maximum concentration was less than 2x the standard, and the 95% UCLM was less than the standard.	Moderate. The total concentrations of metals in soil is not particularly informative for the overall risk conclusion beyond providing a measure of potential exposure. There has been a substantial amount of sampling conducted which provides confidence that the magnitude and extent of exposure is well understood.
Are those bulk concentrations present in a form that could be taken up by plants?	Qualitative. Determine if tailings material is likely to be acid-generating (more likely to leach metals). Qualitative. Determine the magnitude of leachability of Cu and V by measuring the water soluble fraction in field-collected samples and from leach testing.	Low. Determining the bioavailable fraction helps to place the hazard quotients for soil chemistry in context, but does not provide direct measurement of an ecologically relevant effect to plants.	Low. Copper and vanadium were not bioavailable. Water soluble fractions were less than 0.1% of the total metal concentrations. A geochemical conceptual model predicted that copper would have low leachability based on the properties of the tailings which was verified by the leach testing. Material is not acid-generating.	High. There were multiple chemical surrogates that showed that copper and other metals were unlikely to be bioavailable to plants. The site-specific data was consistent with the underlying geochemical understanding of the material.
Are metals actually being accumulated by plants?	Quantitative. Comparison of Cu and V concentrations in field-collected samples to reference (and pre-mining baseline, where available)	Moderate. An increased concentration of metals in tissue samples shows that bioaccumulation is occurring, but does not provide evidence that an adverse effect is likely to occur unless a defensible critical body residue concentration is available.	Moderate for copper. Concentrations in grass and shrubs were statistically different and greater than the mean plus 2SD of the reference. Concentrations in berries and conifers were not. Copper concentrations were dependent on soil concentrations. Low for vanadium. All four plant types had concentrations that were statistically different and greater than the mean 2SD of the reference; however, the uptake by plants did not appear to be related to soil concentrations.	Moderate. There was an adequate sample size to make statistical comparisons, as well as location-specific soil sampling that allowed evaluation of the underlying bioaccumulation relationships.
	Qualitative. Tissue concentration were measured in roots/shoots of grass and pine seedlings exposed to a dilution series of tailings mixed with clean sand	Low. Same discussion as noted above, but with the additional issue that bioaccumulation in the laboratory is not necessarily indicative of the bioaccumulation under field conditions.	Moderate for copper. Copper concentrations in plant tissues did not appear to be correlated to soil concentrations at the range of soil concentrations that is representative of field conditions.	Low. Acts as supporting information for other lines of evidence that showed that the bioavailability of copper in soil appears to be low.

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there physical characteristics that could limit plants in the absence of elevated metals?	Semi quantitative. Comparison of site-specific bulk density values to a literature threshold derived from greenhouse or laboratory-based studies.	Moderate. Reduction in root growth as a result of high bulk density is linked to the assessment endpoint, but the relationship established in a greenhouse or laboratory setting may not be directly transferable to field conditions	Moderate. Bulk density was available for a representative set of field collected samples, and the average bulk density was only marginally better than the threshold value. The bulk density of all tailings was substantially higher than the average measured in reference soils.	Moderate. The mechanism that a dense sample would have small pores that prevent successful root growth is well understood.
	Qualitative. Comparison of site-specific plant nutrient concentrations to case studies from the literature that show that plant growth in tailings can be limited by nutrient deficiencies in the absence of contamination.	Moderate to high. Case study involved fertilization of kimberlite tailings in northern Canada. Case study looked at multiple species over a two year field experiment and measured effects relevant to the assessment endpoint.	Moderate. Kimberlite tailings had similar characteristics as tailings from the site without metal contamination. Case study showed that material similar to the tailings at the site would limit plant growth because of nutrient deficiencies. Amendment with organic carbon was beneficial to long-term plant growth, especially when combined with physical improvement of the soil structure.	High. Provides relevant, long-term experimental validation that tailings are likely to cause some negative effects on plant growth because of nutrient and structural deficiency in the absence of metal contamination.
Do soils exert adverse effects to plants under laboratory conditions?	Quantitative. Compared toxicity test endpoints during 42-d laboratory test on field collected samples to a 20% threshold relative to an artificial negative control.	Moderate. Reductions in seedling germination, emergence, and shoot/root growth are linked to the assessment endpoint, but the relationship established in a greenhouse or laboratory setting may not be directly transferable to field conditions.	Moderate. The samples demonstrated more than a 20% response for pine root growth, as well as grass root and shoot growth endpoints. There was not a 20% reduction in seedling germination or emergence for either species, or for pine shoot growth.	Moderate to high. Toxicity testing showed effects on plants, even though nutrients were not deficient. The 100% samples were also run as a side-by-side dilution series (spiked and non-spiked) that showed that copper bioavailability is lower in tailings than in a spiked study. Structural limitations of the samples (e.g., crusting, poor water retention) are still influencing plant growth.
Is there evidence of adverse effects to the plant community under field conditions?	Semi-quantitative. Field measurements of a wide variety of endpoints (e.g., percent cover; diversity) in field plots. Data compared between years, and relative to reference plots. Differences were evaluated in terms of statistical significance.	High. Provides direct measurement of community-level endpoints that reflect the assessment endpoint.	High. Statistical analysis was only practical for a subset of field endpoints because of difference in field methods between years. However, even a qualitative assessment is sufficient that there was nearly a complete loss of ecological function in the impacted areas as a result of the original tailings deposition. There are multiple indicators that recovery of ecosystem function is starting to occur, but quantitative assessment of the data currently available is not appropriate at this early stage of recovery because of the variability in both species- and ecosystem-level endpoints.	High. The risk assessment was explicitly designed around the observation that a landscape-level, significant impact to the plant community has occurred. The majority of lines of evidence are intended to determine if soil contamination will be an additional barrier to the long-term recovery of ecosystem function. There is no doubt that impacts to the plant community have occurred.

3.5.4 Soil Invertebrates

There were fewer soil invertebrate-specific lines of evidence than there were for plants. Golder notes that having fewer lines of evidence with respect to the soil invertebrate adds uncertainty to the risk assessment. However, it is relevant to consider that the re-establishment of a functional soil invertebrate community would not be expected at this point because of the physical limitations of the soil, and site investigation and rehabilitation efforts have therefore focused on the plant community.

The weight of evidence conclusion was that the presence of residual metals in soil are not likely to prevent reintroduction of a biologically diverse, functional, self-sustaining and inter-dependent soil invertebrate community. This conclusion was developed based on consideration of the plant-specific lines of evidence, as well as those lines of evidence that focused on soil invertebrates. Golder acknowledges that bioaccumulation of copper and vanadium by soil invertebrates is occurring, based on the limited data available. The patterns of copper and vanadium uptake appear consistent with the data obtained for plants. Verification of uptake rates through experimental manipulation or extensive quantitative field measurement has not been conducted. This level of verification would appear to be unnecessary to making an informed risk conclusion at this time. There is every reason to expect that long-term effects to the soil invertebrate community will persist until the underlying physical and nutrient deficiencies are mitigated through rehabilitation and natural ecological succession, and therefore, the soil invertebrate community should be included in the site monitoring program.

A summary of each individual line of evidence with respect to soil invertebrates is provided in Table 23 and the following information provides a narrative overview of how these various lines of evidence were integrated into the overall weight of evidence conclusion. The main lines of evidence related to low bioavailability of metals (e.g., geochemical modelling, leach and other testing) that were previously discussed for plants apply equally to soil invertebrates. There is no scientific reason why metals that have low bioavailability to plants would have high bioavailability to soil invertebrates.

Table 23: Summary of Lines of Evidence for Soil Invertebrates

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there bulk concentrations of metals that are hazardous in soils?	Quantitative – Calculate hazard quotient by comparing soil chemistry to CSR numerical standards for protection of soil invertebrates and plants (or generic standards).	Low. A numerical standard for total metal based on toxicological literature does not account for site-specific bioavailability. The exceedance of a numerical standard does not mean that adverse effects to organisms is actually occurring. The linkage to the assessment endpoint is extremely low for vanadium because it is a generic standard, not a pathway-specific standard.	High for copper. The maximum soil concentrations are up to 10x the standard, and the 95% upper confidence of the mean was approximately 5x the standard. Low for vanadium. The maximum concentration was less than 2x the standard, and the 95% UCLM was less than the standard.	Moderate. The total concentrations of metals in soil is not particularly informative for the overall risk conclusion beyond providing a measure of potential exposure. There has been a substantial amount of sampling conducted which provides confidence that the magnitude and extent of exposure is well understood.
Are those bulk concentrations present in a form that could be taken up by soil invertebrates?	Qualitative. Determine if tailings material is likely to be acid-generating (more likely to leach metals). Qualitative. Determine the magnitude of leachability of Cu and V by measuring the water soluble fraction in field-collected samples and from leach testing.	Low. Determining the bioavailable fraction helps to place the hazard quotients for soil chemistry in context, but does not provide direct measurement of an ecologically relevant effect to plants.	Low. Copper and vanadium were not bioavailable. Water soluble fractions were less than 0.1% of the total metal concentrations. A geochemical conceptual model predicted that copper would have low leachability based on the properties of the tailings which was verified by the leach testing. Material is not acid-generating.	High. There were multiple chemical surrogates that showed that copper and other metals were unlikely to be bioavailable to plants. The site-specific data was consistent with the underlying geochemical understanding of the material.
Are metals actually being accumulated by soil invertebrates?	Qualitative. Comparison of Cu and V concentrations in field-collected samples to reference. There was insufficient sample size to allow for statistical comparisons.	Moderate. An increased concentration of metals in tissue samples shows that bioaccumulation is occurring, but does not provide evidence that an adverse effect is likely to occur unless a defensible critical body residue concentration is available.	Moderate for copper. Concentrations in invertebrates appeared to be higher in exposed areas, but data were limited. Copper concentrations appeared dependent on soil concentrations. Copper concentrations in the small number of earthworm samples were approaching threshold values for where adverse effects were noted in the literature. Low for vanadium. There were no visible patterns and uptake by invertebrates did not appear to be related to soil concentrations.	Low to moderate. There was a limited sample size. The threshold value from the literature may not be directly applicable because of differences in bioavailability.
Are there physical characteristics that could limit invertebrates in the absence of elevated metals?	No specific line of evidence			

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Do soils exert adverse effects to invertebrates under laboratory conditions?	No specific line of evidence			
Is there evidence of adverse effects to the invertebrate community under field conditions?	Observational. Field biologists noted that soil invertebrate communities in exposed areas were altered, but beetles, ants and spiders have been observed in the impacted areas.	Low. Site observations are informative, but are not quantitative or semi-quantitative data that provides direct measurement of community-level endpoints (e.g., diversity, abundance) that reflect the assessment endpoint.	High. The abundance of organisms was low, and hindered the ability to collect sufficient sample sizes and volumes for other lines of evidence.	High. The risk assessment was explicitly designed around the observation that a landscape-level, significant impact to the soil invertebrate community has occurred. The majority of lines of evidence are intended to determine if soil contamination will be an additional barrier to the long-term recovery of ecosystem function. There is no doubt that physical impacts to the soil invertebrates have occurred.

3.5.5 Soil Microbes

The weight of evidence conclusion was that the presence of residual metals in soil are not likely to prevent reintroduction of a biologically diverse, functional, self-sustaining and inter-dependent soil microbial community.

Direct assessment of exposure and effects in the context for protecting the soil microbe community was limited to the 2015 soil ecosystem survey. This survey showed that the root-fungal community in the impacted areas had more saprophytic hyphae than background, and that ectomycorrhizae were generally absent. Saphrophytic hyphae are associated with elevated decay symptomatic of an “unhealthy” ecosystem while ectomycorrhizae are a positive indicator of a “healthy” ecosystem. Physical and chemical variables measured in the field were also consistent with the conclusion that the soil community in the halo area was “unhealthy”, but there were multiple indicators of improvement in these variables in 2016. This included an improvement in the depth of the rooting zone used by plants.

The presence of a growing root system was an important sign of recovery because it is inherently associated with a more robust soil foodweb community (including fungi, bacteria, and higher soil trophic levels) which would contribute to increased aeration, nutrient availability, metal sequestration, and an overall increase in productivity in the tailings/soil mixture. There was no evidence that metals were causing effects to the root system, but the design of the 2016 soil monitoring program was not explicitly designed to detect subtle differences over time, or to distinguish between the structural and nutrient deficiency versus contamination. A study to detect subtle differences would appear to be unnecessary at this time because rehabilitation efforts to improve the structural and nutrient deficiencies are already underway. Effects attributable to the known physical and structural deficiencies are expected to be substantially larger than effects related to copper in soil (if any) given the multiple lines of evidence that show copper in soil has low bioavailability.

3.5.6 Mammals and Birds

There are low risks to mammals and birds as result of copper and vanadium accumulating in the food chain. This risk conclusion was based on the findings from a mechanistic food chain model that was constructed using default parameters established by regulatory guidance, along with site-specific information about metal concentrations in soil, water and dietary items. The mechanistic food chain model achieved a defensible balance between using site-specific data to improve ecological realism while retaining conservative assumptions that reduce the chance of making an incorrect conclusion. There is no weight of evidence for evaluating risks to wildlife—numerical standards for soil are not designed to directly evaluate this pathway. There was no direct measurement of tissue concentrations in mammals and birds or experimental validation of effects under field conditions given the need for destructive sampling and logistical limitations of these approaches. These other lines of evidence with respect to risks to mammals and birds were not considered necessary in light of the low magnitude of hazards calculated by the food chain model.

Table 24: Summary of Lines of Evidence for Mammals and Birds

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there bulk concentrations of metals that are hazardous in soils?	Quantitative – Calculate hazard quotient by comparing soil chemistry to CSR numerical standards for protection of soil invertebrates and plants (or generic standards).	Very Low. A numerical standard for total metal based on toxicological literature for soil invertebrates and plants does not account for site-specific bioavailability and does not have direct relevance to mammals or birds.	High for copper. The maximum soil concentrations are up to 10x the standard, and the 95% upper confidence of the mean was approximately 5x the standard. Low for vanadium. The maximum concentration was less than 2x the standard, and the 95% UCLM was less than the standard.	Low. There was a robust sampling plan to characterize soil quality at the site, but the soil standards are not explicitly based on protection of small mammals and birds.
Are those bulk concentrations present in a form that could be taken up by mammals and birds?	Qualitative discussion of PBET findings.	Low. PBET provides a measure of the fraction of the ingested dose that is potentially available for uptake. PBET does not measure the fraction of the ingested dose that is ultimately absorbed, distributed in tissues and capable of exerting an effect	Low for vanadium. Less than 5% of the total vanadium in soil would be expected to be absorbed in the gastro-intestinal tract. Moderate for copper. Less than 50% of the total copper in soil would be expected to be absorbed in the gastro-intestinal tract.	High. There was sufficient sample size and PBET is an established method for estimating bioavailabity of ingested soil.
Are there hazards to wildlife as a result of the ingestion of food, soil and water?	Hazard quotient: mechanistic food chain model used to compare total ingested dose to toxicity reference value	Moderate. Food chain models are a standard approach, but ultimately make two major inferences. Reductions in growth or reproduction as a result of a contaminant dose in the laboratory are inferred to represent a similar reduction in performance in wild animals in the field. Minor reductions in endpoint performance are inferred to be ecologically relevant to the population.	Low. Hazard quotients were less than one for vanadium for all combinations. Spatially-weighted hazard quotients were less than 1 for copper for all receptors except robin, shrew, flycatcher and nighthawk. The spatially-weighted hazard quotients for those receptors were less than 3. The Polley Flats subarea (untreated) had the highest area-specific hazard quotients for those four receptors, with hazard quotients of up to approximately 5.	High. Food chain models incorporate numerous conservative assumptions. The potential toxicity of ingested vanadium and copper is relatively well understood and is based on a comprehensive literature search conducted by USEPA.

3.5.7 Amphibians and Reptiles

A weight of evidence evaluation has not been completed for amphibians and reptiles at this time. Risk assessments for amphibians and reptiles are often deferred until there is evidence that the substances of interest are accumulating in food chains. It is not correct to conclude that risk assessments for these receptors cannot be completed because of a lack of standardized approaches; however, it is true that the risk assessment must carefully weigh the likely impacts to the population from habitat-related factors as well as classic contaminant-related factors. Risks to amphibians are particularly challenging to evaluate since they occupy the interface between soil and water ecosystems, and have varying sensitivity to contaminants as well as other stressors. A simplistic approach where risks to amphibians or reptiles are inferred based on risks (or lack thereof) to small mammals, benthic organisms or fish is not likely to provide a defensible conclusion except in the limited circumstance where risks to all three groups of organisms are clearly negligible, and then only when the relative sensitivity of amphibians to the specific COPCs are understood.

Qualitatively, a substantial number of amphibians have been observed (and salvaged) from standing water that accumulated in depressions in tailing-rich areas along the Hazeltine corridor. These observations were made over two seasons, suggesting that some degree of successful growth and reproduction is occurring despite exposure to representative worst-case conditions for both sediment and surface water. There were also multiple lines of evidence that suggested that the bioavailability of copper in tailings is limited. Conversely, site-specific data does suggest that copper can bioaccumulate in soil invertebrates to some degree, and an opportunistic tissue sample (n = 1) suggests that bioaccumulation to amphibians is a plausible exposure pathway. No opinion with respect to long-term risks associated with copper or other metals to amphibians are being made at this time.

3.5.8 Implication of Spatial Scale on Risk Conclusions

The preceding section provided a summary of the risk conclusions based on the magnitude of observed response relative to the decision criteria for individual lines of evidence. The evaluation focused on describing how different lines of evidence were contributing to the overall risk to the ecosystem on a site-wide basis. An uncertainty analysis was conducted for each individual line of evidence (and reported in preceding sections). The overall uncertainty in the line of evidence reflected the degree to which it was used to inform the risk assessment conclusion. This type of uncertainty analysis is routine in an ecological risk assessment. However, in addition to the uncertainty associated with individual lines of evidence, there is an additional consideration for risk management in how risk conclusions based on the site as a whole can be used to inform site management planning at a less than site-wide basis. The following section provides a discussion of this specific issue to assist risk managers.

3.5.8.1 Spatial Interpolation of Individual Lines of Evidence

A practical example of how spatial scale can influence risk management decisions relates to wildlife receptors that have a small home range. On a site-wide basis, the risk assessment assumes all available habitat will be used on an equal basis, that receptors are mobile, and therefore, exposure will reflect that tendency for averaging (which is conservatively expressed in the risk assessment as a 90th percentile or 95% upper confidence limit of the mean). The protection goal in the risk assessment for a common wildlife species is to protect the population, not specific individuals, but from a practical risk management perspective, it may still be desirable to focus risk management actions onto specific areas where a small proportion of a given wildlife population may in fact experience a higher than average exposure. This risk management question does not arise for most contaminated sites because the total area of impact is often only a minor portion of a typical habitat range. In this specific risk assessment, the extent of the impact is large, and therefore, the ability to interpolate the overall risk conclusion to subareas to support risk management planning may be relevant.

Each line of evidence varies in terms of its ability to support interpolation to a sub-area level to help support informed risk management decisions. Table 25 provides an overview of each line of evidence in terms of its interpolative ability.

Table 25: Spatial Scale of Different Lines of Evidence

Line of Evidence	Decision Criterion	Spatial Coverage	Ability to Interpolate to a Subarea
Are there bulk concentrations of metals that are hazardous in soils?	Calculate hazard quotient by comparing soil chemistry to CSR numerical standards for protection of soil invertebrates and plants (or generic standards).	High. Soil chemistry data collected from variety of subareas with good spatial coverage of the site as a whole.	High. Hazard quotients can be calculated for individual samples where appropriate.
Are those bulk concentrations present in a form that could be taken up by plants or soil invertebrates?	Determine the magnitude of leachability of Cu and V by measuring the water soluble fraction in field-collected samples and from leach testing.	Low for leach testing. Leach testing was conducted on representative worst-case samples of tailings and was not intended to provide spatial coverage. Moderate for water soluble fraction. The water soluble fraction (paste extracts) was measured on a subset of samples collected from different treatment areas.	Moderate to high. Leach testing showed low bioavailability in a representative worst-case tailings. Paste extract data was collected from soils with a range of copper and vanadium concentrations.
Are those bulk concentrations present in a form that could be taken up by mammals and birds?	Discussion of PBET findings.	Moderate. PBET analysis was conducted on a subset of samples from different treatment areas.	Low to moderate. There was insufficient data to evaluate a relationship between total metals and bioaccessible metal concentrations.
Are there physical characteristics that could limit plants or invertebrates in the absence of elevated metals?	Determine bulk density and plant available nutrients	Low for bulk density. Bulk density was measured on a subset of nine tailings samples. Moderate for plant available nutrients. Plant available nutrients were measured on a subset of samples taken from different treatment areas.	Moderate to high. Bulk density measurements on tailings provide a representative worst case estimate. Plant available nutrients were collected from a variety of different treatment areas.
Are metals actually being accumulated by plants?	Comparison of Cu and V concentrations in field-collected samples to reference (and pre-mining baseline, where available)	Moderate. Grass and shrub samples were collected from the halo area and the floodplain. Conifer and berry samples were limited to the halo area only.	High. Tissue samples were collected over a range of soil concentrations. The relationship between soil and plant concentrations was evaluated.
Are metals actually being accumulated by soil invertebrates?	Comparison of Cu and V concentrations in field-collected samples to reference. There was insufficient sample size to allow for statistical comparisons.	Low. There were limited soil invertebrates within the floodplain for sampling.	Low. There was insufficient data to establish relationships between soil and invertebrate concentrations.
Do soils exert adverse effects to plants under laboratory conditions?	Compared toxicity test endpoints during 42-d laboratory test on field collected samples to a 20% threshold relative to an artificial negative control.	Low. Toxicity testing was conducted using dilutions of two representative tailing samples. Spiked study was also conducted with inert sands and silts as well as a greenhouse soil control.	Low to moderate. Toxicity testing was not intended to simulate different soil treatment options.
Is there evidence of adverse effects to the overall soil community under field conditions?	Field assessment of forest health in halo area.	Moderate. Assessments were conducted at numerous locations within the halo zone over two different years.	Moderate. No areas within the floodplain were assessed using this tool.
Are there hazards to wildlife as a result of the ingestion of food, soil and water?	Hazard quotient: mechanistic food chain model used to compare total ingested dose to toxicity reference value	High. Food chain models were run on different treatment areas as well as spatial distinct portions of the untreated area.	High. The food chain model incorporates exposure data which is generally tied to the soil concentrations.

3.5.8.2 Differences in Soil Chemistry

The fundamental question for interpolating the findings of the terrestrial risk assessment to specific subareas relates to the amount of tailings present in any given area. Areas with a high proportion of tailings will tend to have higher metal concentrations, but also a physical structure that lacks organic carbon. Conversely, areas with a low proportion of tailings tend to have lower metal concentrations but also a more natural soil structure that facilitates plant growth and reestablishment of the soil invertebrate communities. This gradient was considered in the risk assessment design in a variety of ways. At a broad level, the actual magnitude of change in metals concentrations along this exposure gradient is relatively small. Table 26 provides a breakdown of the site-wide soil chemistry data set into specific subareas evaluated by the wildlife food chain model. The subareas are highlighted in Figure The current focus of the site rehabilitation continues to be to intermix the Polley Flats area.

Table 26: Summary of Soil Concentrations along Spatial Gradient

Spatial Gradient	N	Copper (95th UCLM mg/kg)	Vanadium (95th UCLM mg/kg)
Untreated	182	737	177
Polley Flats	*44*	*931*	*200*
Upper Hazeltine Corridor	*77*	*738*	*181*
Lower Hazeltine Corridor	*61*	*656*	*163*
Treated - Transplanted	30	197	96
Treated - Intermixed	37	601	150
Treated - Scarified	23	247	99

3.5.8.3 Conclusion

Golder concludes that the overall risk conclusions described in Section 3.5.2 on a site-wide basis can be broadly applied to different portions of the site. The risk assessment focused on a conceptual site model that the impacts are associated with the deposition of tailings (which are relatively uniform in composition) albeit over a relatively large area. Some lines of evidence considered in the risk assessment focused on “halo” versus “floodplain” (i.e., two representative areas along the gradient) while others focused on a worst-case “tailings” scenario. Overall, the minor differences in exposure concentrations between the Polley Flats and other portions of the Hazeltine Channel are not considered likely to result in major differences in the overall risk conclusion beyond what has already been described in the risk assessment, especially given that the worst-case soil conditions in the Polley Flats area continue to be the focus for rehabilitation.

LEGEND

HAZELTINE CORRIDOR AREAS
- POLLEY FLATS (UNTREATED PORTION 50ha)
- UPPER HAZELTINE (UNTREATED PORTION 35ha)
- LOWER HAZELTINE (UNTREATED PORTION 35ha)

RESTORATION TYPE
- INTERMIXED SOIL ZONE (49ha)
- SCARIFIED ZONE (33ha)
- TRANSPLANTED SOIL ZONE (9ha)

BASEDATA
- HAZELTINE CREEK CHANNEL (APPROXIMATE)
- ROAD
- WATERCOURSE

NOTE
1. UNSHADED AREAS WITHIN THE HAZELTINE CORRIDOR ARE UNTREATED. THE POLYGONS REPRESENT RESTORATION UP TO NOVEMBER 2016.

0 575 1,150
1:24,000 METRES

REFERENCES
1. ROAD AND WATER COURSE DATA OBTAINED FROM OBTAINED FROM GEOGRATIS, © DEPARTMENT OF NATURAL RESOURCES CANADA. ALL RIGHTS RESERVED.
2. IMAGERY OBTAINED FROM MPMC, MAY 2015
3. BACKGROUND IMAGERY COPYRIGHT © 20161201 ESRI AND ITS LICENSORS. SOURCE: DIGITALGLOBE. USED UNDER LICENSE, ALL RIGHTS RESERVED. IMAGE DATE: 20101104

4. DATUM: NAD83 PROJECTION: UTM10

CLIENT
MOUNT POLLEY MINING CORPORATION
IMPERIAL METALS

PROJECT
MOUNT POLLEY ECOLOGICAL RISK ASSESSMENT

CONSULTANT
Golder Associates

YYYY-MM-DD	2017-09-21
DESIGNED	EZG
PREPARED	CD
REVIEWED	EZG
APPROVED	AA

TITLE
LOCATION OF SUBAREAS FOR TERRESTRIAL RISK MANAGEMENT

PROJECT NO.	PHASE/DOC	REV.	FIGURE
1662612	22300	0	19

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4.0 AQUATIC ECOLOGICAL RISK ASSESSMENT

4.1 Introduction

4.1.1 Review of the Problem Formulation

Section 1.1 highlighted that the purpose of the aquatic risk assessment is to determine if the contaminant stressor (i.e., residual concentrations of metals in tailings) represents an unacceptable risk that requires risk management over and above the rehabilitation (in the Hazeltine Channel) and monitored recovery program (in Quesnel and Polley Lakes) that is already underway to address the physical stressors inherent from the scouring and deposition of tailings. The management goal is to achieve the maintenance (or successful reintroduction) of a biologically diverse, functional, self-sustaining and inter-dependent ecosystem. The available site information was reviewed to identify COPCs, receptors of potential concern and viable exposure pathways. Based on this information, the following assessment endpoints and major lines of evidence were articulated:

- Assessment Endpoint 1: Protect aquatic receptors from potentially hazardous metal exposures via direct contact with water and sediment. Lines of evidence related to this assessment endpoint focus on comparing the total concentrations of COPCs to risk-based numerical guidelines, standards, or screening values, as well as surrogate measurements of how much of the total concentration is likely to be mobile or bioavailable (e.g., leaching potential from tailings; sequential extractions; bioavailability models such as AVS-SEM).
- Assessment Endpoint 2: Protect benthic organisms from impairment of functional health attributes (e.g., survival, growth, reproduction, or normal development) as a result of direct contact with sediment. Lines of evidence related to this assessment endpoint include measurements of effects in the field or in the laboratory, or provide a direct evaluation of bioavailability (e.g., diversity and abundance of resident benthic organisms, toxicological effects in standardized laboratory toxicity tests, tissue bioaccumulation comparisons between exposed and reference areas.
- Assessment Endpoint 3: Protect plankton from reductions in survival, growth, and reproduction as a result of direct contact with water influenced by tailings. This assessment endpoint is similar to the one described above for benthic organisms and incorporates lines of evidence such as direct measurement of zooplankton diversity and abundance and phytoplankton biomass in the field, as well as measurements of adverse effects in the laboratory using standardized toxicity tests.
- Assessment Endpoint 4: Protect fish from reductions in impairment of functional health attributes (e.g., survival, growth, reproduction, or normal development) as a result of direct contact with abiotic media or ingestion of prey items. This assessment endpoint includes several lines of evidence intended to determine if adverse effects are occurring to individual fish or to fish populations. Lines of evidence related to this assessment endpoint include consideration of metals concentrations in prey items, concentrations bioaccumulated into fish tissues (i.e., the sum of exposure from prey items and water), standardized toxicity tests, direct measures of fish disease (histopathology), and direct measures of individual and population condition (e.g., condition factor and demographics).

4.1.2 Content of this Chapter

This chapter provides a review of each individual line of evidence that was considered in the exposure and effects assessment stages of the aquatic risk assessment. This review provides:

- A brief discussion of the study design and methodology used for data collection (including consideration of how the line of evidence met its underlying data quality objectives)
- A summary of the results obtained for the line of evidence
- A discussion of how the decision criterion for the line of evidence were developed and presentation of how the resultant study data compare to those decision criteria
- A discussion of the uncertainty in the data and their interpretation following the decision criterion
- A summary of the key findings for use in the weight-of-evidence integration

The technical reports as prepared by the principal investigators for each study component are provided as appendices; these reports provide the detailed information on individual samples, specific analytical results, detailed methodologies, and quality assurance. Some technical reports were developed to address specific study questions at the time to address items that the public needed to know or to provide interim recommendations for preliminary risk management decisions, reflecting the overall adaptive nature of the approach used by the mine and by government (Nikl et al., 2016). A key objective of this report section is to aggregate the technical information from multiple rounds of testing, synthesize technical information from many different investigators, and align that information with provincial risk assessment guidance.

4.2 Exposure-Based LOE: Water Chemistry

4.2.1 Water Chemistry

Copper was identified as a COPC in surface water based on chemistry data collected between May and November 2015, as described in the aquatic problem formulation (Section 2.4.2.2). A review of the water chemistry data collected in 2016 indicated there were no additional substances that exceeded water quality guidelines and the background concentrations. Therefore, Golder focused on suspended particulates and copper in the water chemistry line of evidence below.

4.2.1.1 Study Design and Methodology

The assessment of water chemistry relied on data collected by MPMC following the TSF embankment breach. In general, 2015 and 2016 monitoring focused on routine stations, but a small number of stations were moved, renamed, added, or discontinued, consistent with the continued adaptive evolution of the post-breach monitoring program. A number of stations in Polley Lake, the Hazeltine Channel, Quesnel Lake, and Quesnel River were consistently monitored, typically either weekly or monthly. Monitoring included measurement of field parameters at various depths as well as along depth profiles for lakes, and collection of discrete water samples, including at various depths in both lakes. Details of sampling methods and stations are provided in MPMC (2015) and Appendix A-6. Analytical and field monitoring data were verified as reliable based on a quality assurance and quality control (QA/QC) assessment by MPMC and data were provided electronically to Golder by MPMC for use in the water quality assessments.

Suspended particulates and copper were evaluated by examining temporal trends at representative, routine monitoring stations in each of the following areas[29]:

- Five locations in the Hazeltine Channel (HAC) (stations HAC-10, HAC-13, HAC-05/a, HAC-08/a, and HAC-01/a/b/c) to represent conditions in the upper and lower sections of the creek. Monitoring locations and frequency evolved in response to changes in the creek outflow in 2015 and commencement of effluent discharge on 1 December 2015.
- Two locations in Polley Lake (stations P1 and P2) that represent the deepest areas of the lake and provide geographical coverage.
- Four locations in Quesnel Lake (stations QUL-2/a, QUL-66[30], QUL-18, and QUL-120/a) representative of near-field, mid-field, and far-field areas relative to the Hazeltine Channel inputs.
- One location in Quesnel River (QUR) (station QUR-1) at the outflow of Quesnel Lake.

Water quality following the TSF embankment breach and through 2016 was examined by comparing data to applicable maximum and 30-d WQGs. Temporal and spatial trends in concentrations of suspended particulates and copper were examined and differences among surface samples and samples collected at depth in Polley and Quesnel lakes were also considered. Water quality trends for 2015 and 2016 are discussed in the following section. The 2014 data were not included here because water quality was influenced by suspended particulates from the original release of tailings; these results, which were discussed previously in MPMC (2015) are not reflective of present and future conditions.

4.2.1.2 Summary of Findings

Hazeltine Channel and Edney Creek

Turbidity and total suspended solids (TSS) in the Hazeltine Channel showed a progressive decrease from observed levels in the first few months of 2015 and stabilized to levels closer to the respective WQGs following the completion of stabilization and rock armoring on May 11, 2015 (Bronsro et al., 2016; Figure 20; Appendix A-6). Intermittent exceedances occurred in some areas of the creek during freshet of spring 2016 and during remediation activities that were undertaken in upper Hazeltine from July through September 2016. In lower Edney Creek, turbidity and TSS were typically below WQGs (Appendix A-6).

[29] Descriptions of stations and rationale for selection of representative stations are provided in Appendix F of MPMC 2015. See Attached 1 of Appendix A-6.1 for map showing sampling locations.

[30] QUL-66 was replaced by QUL-55, QUL-55a, and then QUL-58 in response to changes in the location of the Hazeltine Channel outflow into the lake. These stations have contiguous sampling periods and for the purposes of this assessment are considered to represent the same location and are collectively referred to as QUL-66 in the following sections.

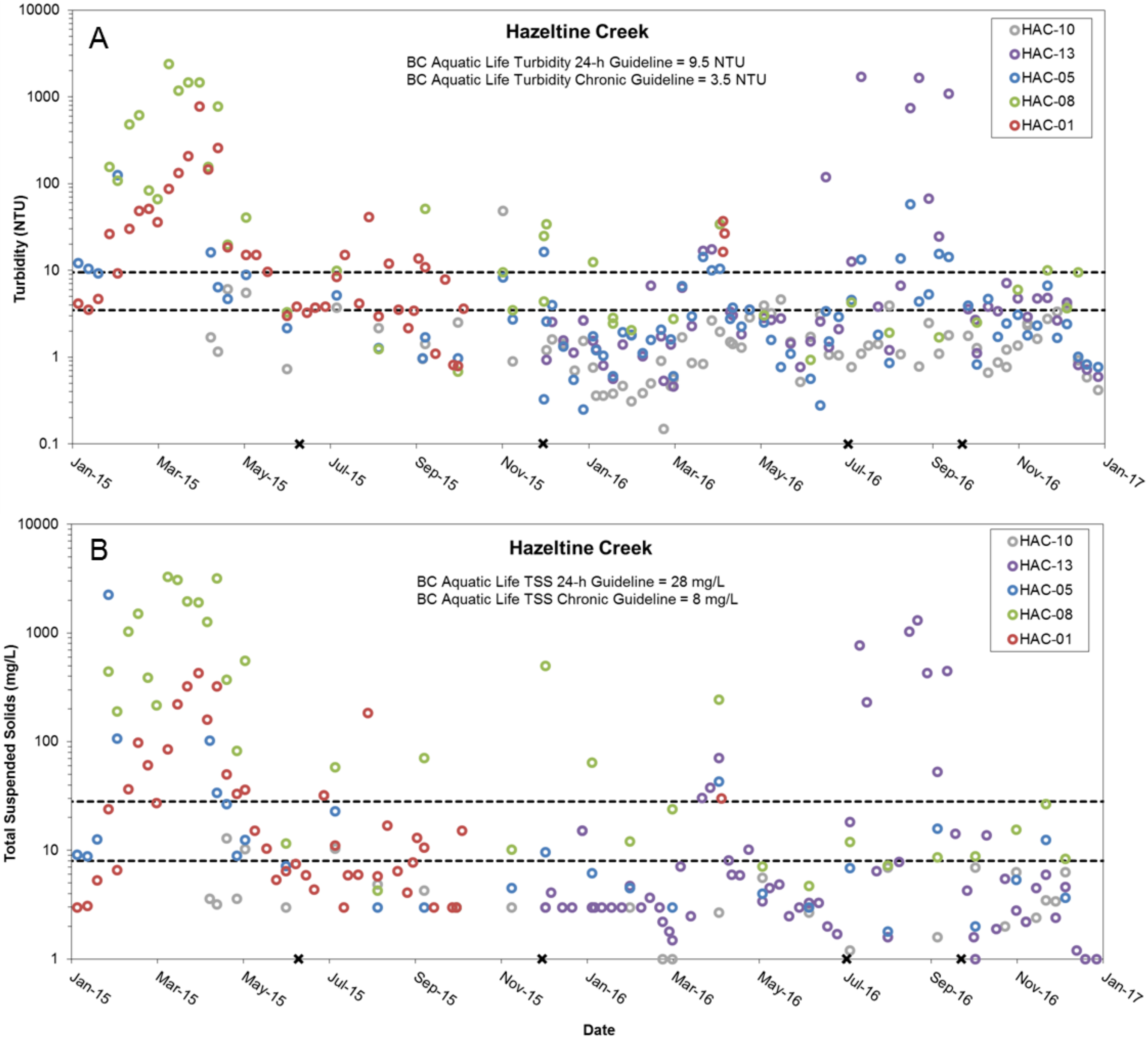

Figure 20: Turbidity (A) and Total Suspended Solids (B) Concentrations at Stations in the Hazeltine Channel, 2015 to 2016

Note: Stations are ordered from upstream (HAC-10) to downstream (HAC-01). Stations HAC-10 and -13 are upstream of the effluent discharge location. '**x**' denotes the following events: May 2015 completion of initial rock armouring, December 2015 commencement of effluent discharge to lower creek, July through September 2016 remediation of upper Hazeltine. Background concentrations of 1.5 NTU and 3 mg/L in the Hazeltine Channel were used for the BC WQGs.

Total and dissolved copper concentrations in the Hazeltine Channel showed a progressive decrease from levels observed in the first few months of 2015 to concentrations closer to WQGs after May 2015 (Figure 21; Appendix A-6). Peaks in copper concentrations corresponded with changes in turbidity and TSS within the creek. Total copper was typically at or below the maximum WQG, but above the chronic WQG through the end of 2016 (Figure 21A). Dissolved concentrations frequently exceeded the 30-d guideline in certain portions of the creek and occasionally exceeded the maximum WQG at some stations (Figure 21B). Golder understands that there may have been further instances since January 2017 where the dissolved copper concentrations in some locations in the Hazeltine channel have exceeded WQG but that management actions have already been executed.

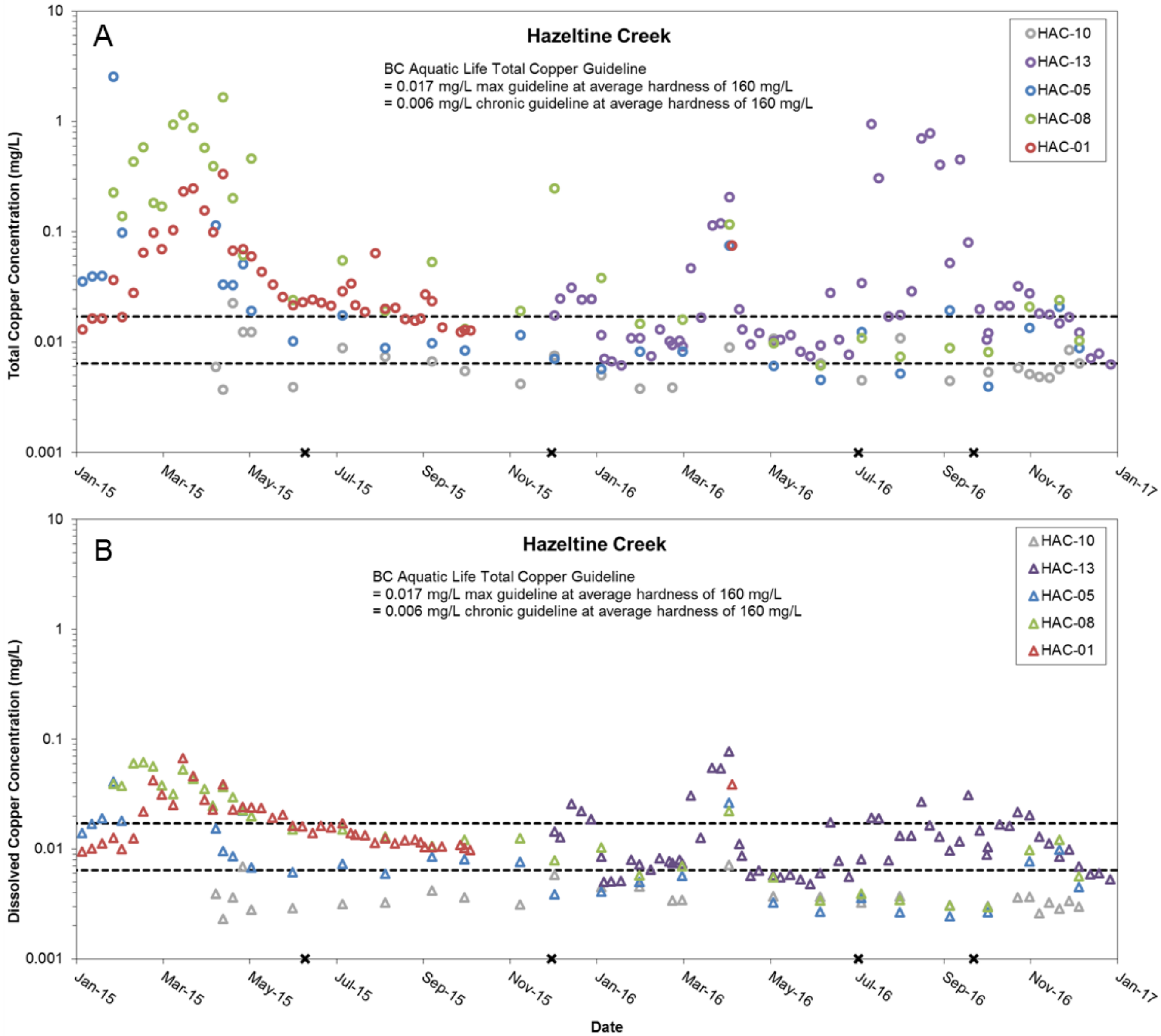

Figure 21: Total (A) and Dissolved (B) Copper Concentrations at Stations in the Hazeltine Channel, 2015 to 2016

Note: Stations are ordered from upstream (HAC-10) to downstream (HAC-01). Stations HAC-10 and -13 are upstream of the effluent discharge location. '**x**' denotes the following events: May 2015 completion of initial rock armouring, December 2015 commencement of effluent discharge to lower creek, July through September 2016 remediation of upper Hazeltine.

Polley Lake

In Polley Lake, no parameters were identified as stressors or COPCs in the problem formulation. Although turbidity (but not TSS) and total copper were initially elevated at depth in Polley Lake after the TSF embankment breach, concentrations decreased over time to below applicable WQGs by fall 2014 as particulate matter settled out of the water column (MPMC 2015). Dissolved copper concentrations were below the WQGs. In 2016, total copper marginally exceeded the 30-d WQG on some occasions, but dissolved copper consistently remained below the WQG (Figure 22).

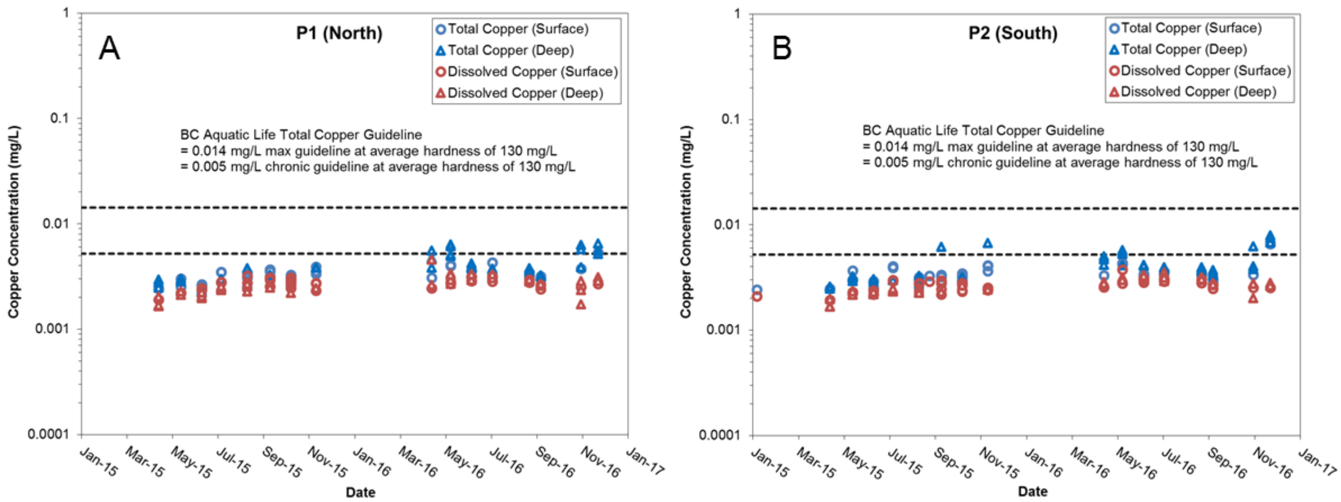

Figure 22: Total and Dissolved Copper Concentrations at Stations P1 (A) and P2 (B) in Polley Lake, 2015 to 2016

Quesnel Lake

In Quesnel Lake, turbidity (but not TSS) and total copper were identified as stressors or COPCs based on chemistry data collected between March and August 2015. After the TSF embankment breach, turbidity was initially elevated at depth in the West Basin, indicative of the plume of suspended and dissolved materials deposited in the lake, but was not elevated at the far-field station east of Cariboo Island (MPMC 2015; Appendix A-6). Turbidity decreased over time as particulate matter settled out and measurements were generally similar throughout the water column following lake turnover in fall 2014. By spring 2015, most turbidity measurements were below WQGs and continued to decline or stabilize through 2015 and 2016 (Figure 23; Appendix A-6)[31].

[31] The increased turbidity at station QUL-18 in July 2016 was considered to be a potential data error caused by equipment (e.g., debris or bubbles), as the same increase was not observed in the laboratory results, or other parts of the lake.

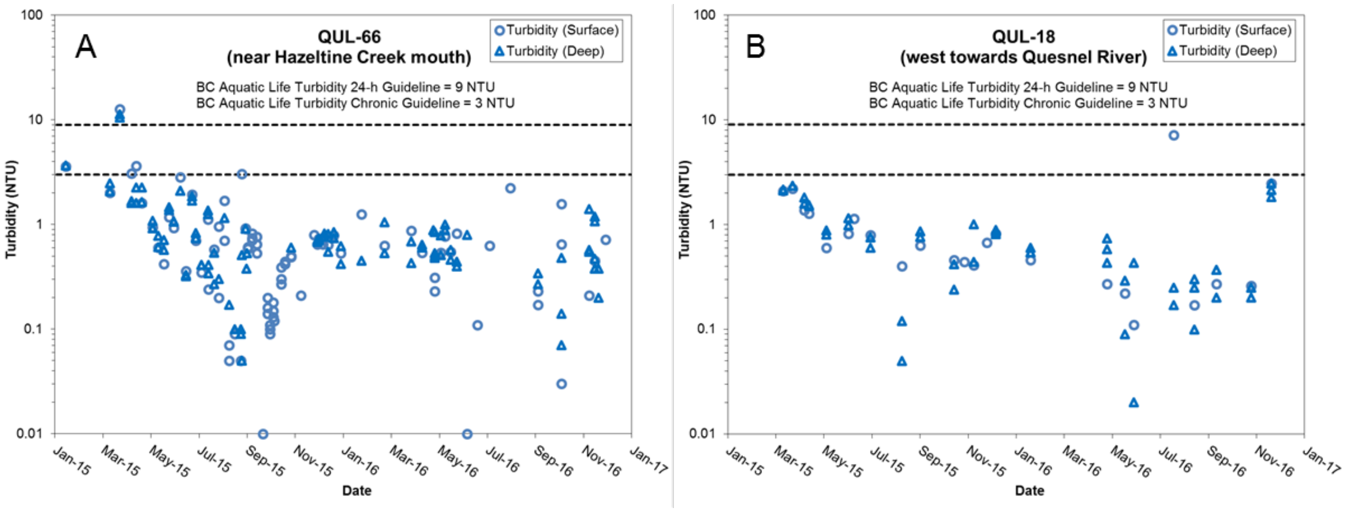

Figure 23: Turbidity at Stations QUL-66 (A) and QUL-18 (B) in Quesnel Lake, 2015 to 2016

Note: Background concentration of 1 NTU in Quesnel Lake was used for the BC WQGs.

Total copper concentrations were elevated at depth in the West Basin of Quesnel Lake after the TSF embankment breach, but not at the far-field station (QUL-120/120a) east of Cariboo Island (Appendix A-6). Dissolved concentrations (surface and depth) and total concentrations at the surface were typically below the maximum WQG. Total copper concentrations at depth decreased over time and were below typically below WQGs by early 2015 with the exception of measurements near the mouth of the Hazeltine Channel (near-field station QUL-66). By late summer 2015, near-field copper concentrations were similar to those elsewhere in the lake, including the east far-field station, and were below WQGs (Appendix A-6). Total and dissolved copper concentrations were below WQGs through the remainder of 2015 and 2016, with the exception of a few samples. WQG exceedances at the near-field station are likely associated with periods when the Hazeltine Channel flows were more turbid and contained higher TSS levels as a result of rehabilitation and construction work during those periods (Figure 24; Appendix A-6).

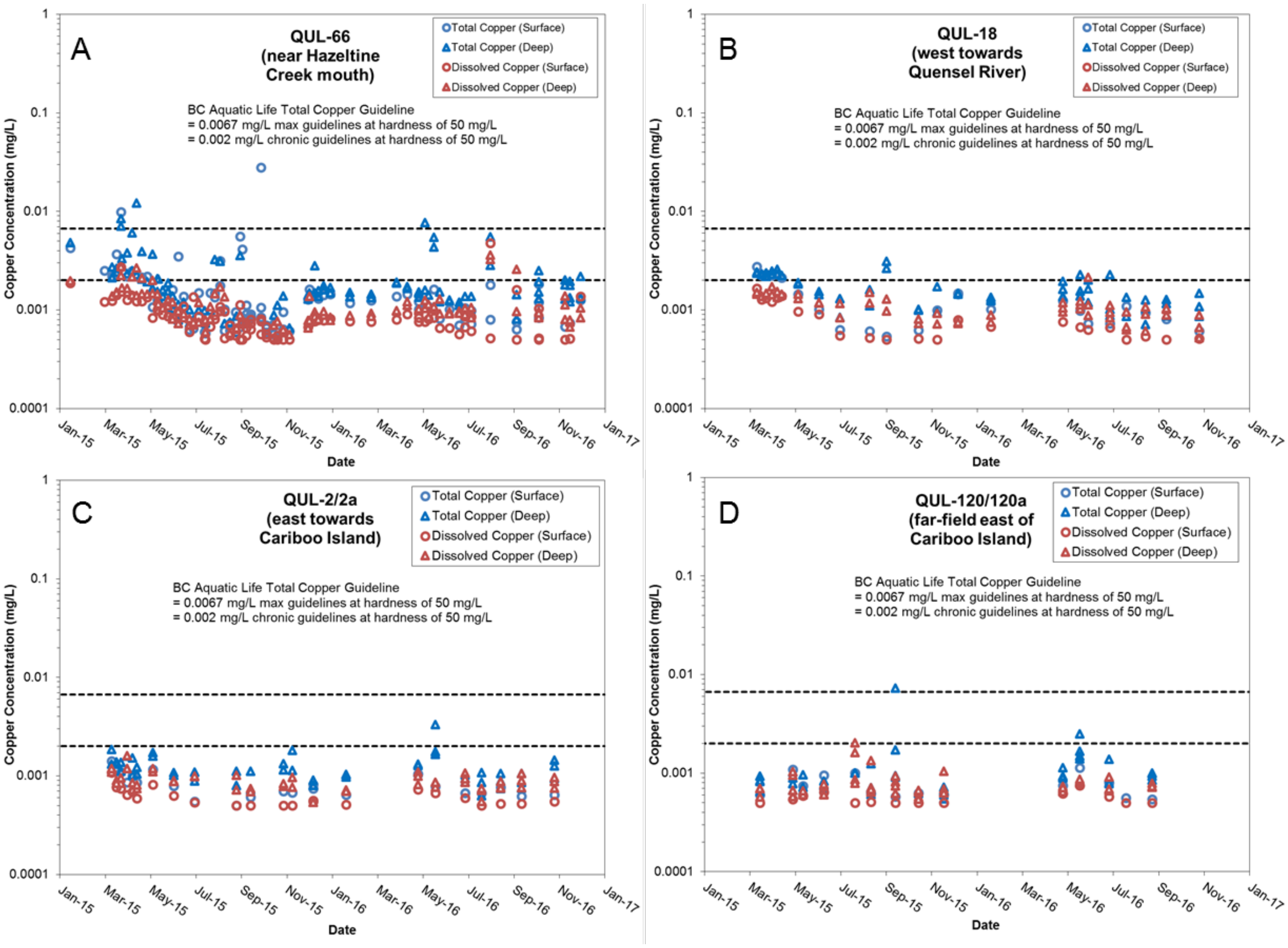

Figure 24: Total and Dissolved Copper Concentrations at Stations QUL-66 (A), QUL-18 (B), QUL-2/2a (C), and QUL-120/120a (D) in Quesnel Lake, 2015–2016

Quesnel River

In Quesnel River, turbidity was identified as a stressor based on in-situ data collected with a continuous logger at station QUR-1 between March and August 2015. Following the TSF embankment breach, turbidity was typically below WQGs except in November 2014 to late January 2015 due to fall turnover of Quesnel Lake and mixing of deep turbid water (MPMC 2015). Daily average and in-situ turbidity subsequently decreased throughout 2015 to levels below WQGs (Appendix A-6). A few intermittent spikes measured by the continuous logger in mid-2015 were observed, but were thought to be influenced by sensor fouling associated with long-term deployment (Appendix A-6) because these same increases were not observed in the in-situ grab samples. Total copper concentrations at station QUR-1 peaked with turbidity from November 2014 to late January 2015, and returned to below WQGs by April 2015 (Appendix A-6). Dissolved copper concentrations did not exceed WQGs.

4.2.1.3 Potential for Copper Bioavailability

The COPC in surface water was copper. Suspended particulates were identified as a potential confounding factor in evaluating the influence of copper because total metal concentrations in water include metals adsorbed or bound to particulate matter, and a correlation often exists between total metal concentrations and turbidity (a surrogate measure for TSS).

Although ambient WQGs are generally intended to be applied to the total form of a metal, the dissolved concentration is known to be a more reliable predictor of toxicity. The copper WQG allows for the form of the copper compared to the WQG to be modified if detailed knowledge regarding bioavailability is available[32]. Evaluations of copper bioavailability in the receiving environment were conducted previously (Appendix F of MPMC 2015) and considered the relationship between turbidity and copper in addition to geochemical speciation modelling.

- **Relationship between Turbidity and Copper**—The correlation between turbidity and total and dissolved copper was examined for the receiving environment. For each of the representative stations, there was a significant positive correlation between total copper concentration and turbidity. In Polley Lake, there was a significant negative correlation between dissolved copper concentration and turbidity; dissolved copper concentrations were below method detection limits in samples with the highest turbidity. In the Hazeltine Channel, there was no relationship between dissolved copper and turbidity, whereas in Quesnel Lake and Quesnel River there was a significant positive correlation between dissolved copper and turbidity (Figure 25; MPMC 2015). These positive relationships suggest that measures of dissolved concentrations may have included fine particulate matter and adsorbed copper that passed through the 0.45-µm filter, particularly in Quesnel Lake (material that passes through a 0.45-µm filter is operationally defined as "dissolved"). Consequently, reported dissolved metal concentrations have likely overestimated the component that would have been bioavailable for direct uptake from the water by aquatic biota.

- **Copper Speciation Modelling**—An understanding of the speciation of the dissolved metal fraction (as opposed to that which is operationally defined as dissolved because it passes through a 0.45-µm filter) provides a truer measure of the bioavailability because it represents the free metal species that are bioavailable for uptake from the water, with the potential to elicit adverse effects (Campbell et al. 2006). Speciation modelling was conducted for copper in Quesnel Lake and Quesnel River using the pH Redox Equilibrium in C language (PHREEQC) computer geochemical equilibrium modelling program (Parkhurst and Appelo 1999). The speciation modelling indicated that the majority of dissolved copper is present as copper carbonate, which is less bioavailable for uptake by aquatic organisms compared to the free ion copper (Cu^{2+}) (MPMC 2015). Therefore, only a small percentage of dissolved copper concentrations is expected to be bioavailable for direct uptake from the water by organisms in Quesnel Lake and Quesnel River. Results of toxicity tests confirmed the low bioavailability of copper, as no toxicity attributable to copper was observed (see Sections 4.4.1 and 4.5.1; MPMC 2015).

[32] BC MoE (1987): Footnote 2 of Table 1 reads "when detailed knowledge on the bioavailable forms of copper is available, the form of copper in the criteria for aquatic life can be modified, as justified by the data"

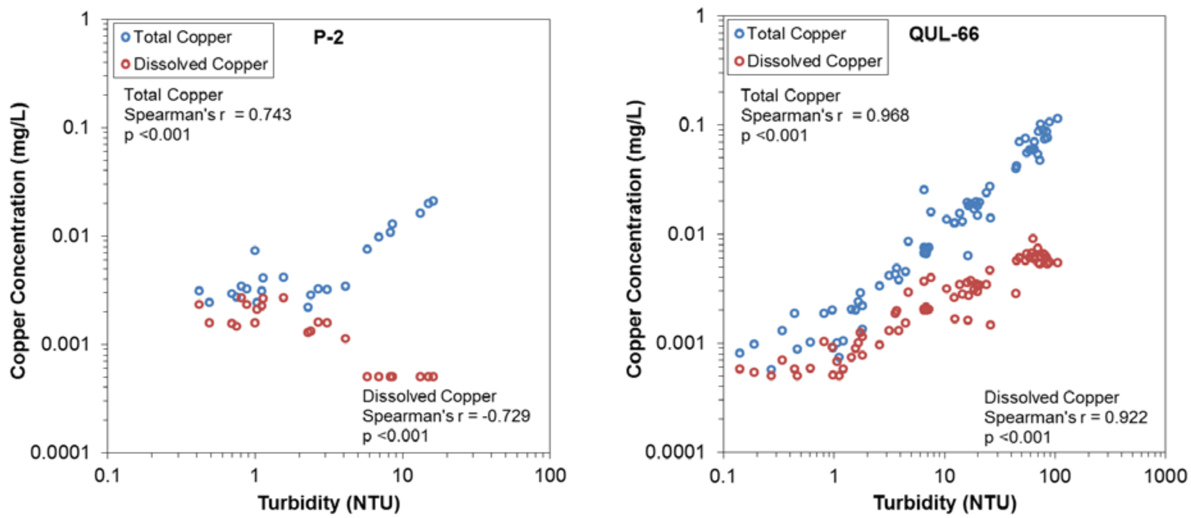

Figure 25: Correlation between Turbidity and Total and Dissolved Copper at Stations P-2 in Polley Lake and QUL-66 in Quesnel Lake (Figure 28 from Appendix F of MPMC 2015)

4.2.1.4 Comparison to Decision Criteria and Uncertainty Analysis

The decision criteria for evaluating the magnitude of the hazard related to copper in surface water relied on comparison of the reasonable worst-case exposure concentrations (i.e., the 95th percentile values of the most recently collected surface water). The analysis focused on the contaminant of potential concern (copper) and a potential physical stressor (turbidity) that had been identified based on a similarly conservative screening of earlier data. Golder used specific windows of time to identify the COPCs, and then to evaluate their long-term hazards against the decision criterion, but ultimately, all data up to January 2017 were presented to provide context about the long-term trends. The long-term trend has been the gradual reduction of turbidity in the various waterbodies since the initial TSF embankment breach. Concentrations of total metals have also declined (and are associated with turbidity), leaving copper as the primary substance for evaluation in the aquatic risk assessment, and then only with respect to the Hazeltine Channel.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Water chemistry	Protect aquatic plants, plankton, and fish from reductions in survival, growth or reproduction resulting from direct contact with water impacted by deposited tailings.	Compare surface water quality to ambient water quality guidelines for the protection of aquatic life.	The 95th percentile concentrations is less than or equal to the applicable guideline or background concentration.	Polley Lake	No hazard
				Hazeltine Channel	Hazard is present
				Quesnel Lake	No hazard
				Quesnel River	No hazard

4.2.2 Diffusive Gradient Thin-Film Passive Samplers

Passive sampling is the general term for environmental sampling using resins or other absorptive materials that provide a more realistic measure of the bioavailable fraction of the COPCs relative to the total concentrations typically measured in sediment or water. The specific device used in the current risk assessment was a diffusive gradient thin-film (DGT) sampler. A DGT sampler consists of a solid plastic or metal cylinder that contains an absorptive resin. The cylinder has “windows” covered with a diffusion membrane that only allows the freely-available portion of metals to pass through for absorption. The freely-available fraction consists of the metal ions as well as metals that are weakly bound to organic carbon and other ions that can easily dissociate back into their ionic form. The DGT sampler is deployed in the environment for several weeks so that that the concentration of metals absorbed by the resin is in equilibrium with the freely-available portion of metals in the surrounding water column. The mass of the metal absorbed in the resin is measured at the end of the exposure period, and can be used to back-calculate the average concentration of freely-available metals in the water column. DGT samplers are recommended for assessing metals (see Peijenburg et al. 2014 for further information about passive samplers in general and specifically DGT).

4.2.2.1 Study Design and Methodology

DGT samplers were deployed by Minnow for approximately 30 days in September 2015 and 35 to 50 days in late August to mid October 2016. Technical reports prepared by Minnow for each of these sampling events are provided in Appendix K-1 and K-2. In brief:

- The sampling devices consisted of a polypropylene cylinder with a dimension of approximately 2 inches, filled with Chelex-100 and Metsorb absorptive resins. The mixture of these two resins was intended to allow the passive sampler to be effective in absorbing metals that tended to be labile (i.e., readily dissociable as divalent ions; Mn, Co, Ni, Cu, Cd and Pb) as well as those metals that tended to form oxy-anionic complexes (i.e., As, Mo, Sb, V). This mixture approach was developed by Panther et al. (2013).
- The devices were purchased pre-loaded with the resin mixture from Griffin University (Brisbane, Australia) by Maxxam Analytics. Maxxam Analytics determined the mass of each metal in the resin mixture in a manner consistent with manufacturer’s recommendations. This included measurement of metal concentrations in fabrication controls (i.e., a sample of resin on arrival at Maxxam), a method blank (i.e., a laboratory blank without resin added) as well as a method spike (i.e., passive sampler placed in a solution of metals of known concentration under laboratory conditions).
- Devices were deployed in 2015 as clusters of four sampling devices in the Hazeltine Channel, Polley Lake, Bootjack Lake (reference), Quesnel Lake (at the mouth of the Hazeltine Channel) and Quesnel Lake (reference). DGT devices were deployed in 2016 as clusters of three sampling devices at stations in the Hazeltine Channel, Edney Creek (reference), Polley Lake, Bootjack Lake (reference), Quesnel Lake (at the mouth of the Hazeltine Channel) and Quesnel Lake (reference). Devices were deployed in 14 areas in 2016 including mid-depth and deep areas in Polley Lake and impacted littoral and profundal areas in Quesnal Lake (Table 1 of Appendix K-2). Field blanks and trip blanks were included. Devices in the lakes were held at a point approximately 1 m above the sediment-water interface. The devices in the Hazeltine Channel were placed in a pool approximately 15 cm above the sediment-water interface. Devices were retrieved after the 30 to 50-day exposure period, rinsed with metal-free water, and transported to the analytical laboratory (Maxxam Analytics, Burnaby, BC). In 2016, detection limits were improved through analysis using Maxxam’s ultra-trace analytical methods.

- Water samples were collected during deployment and retrieval and analyzed by ALS Environmental (Burnaby, BC) for total and dissolved metals concentrations. Maxxam also back-calculated the average "freely-available" water concentration using the manufacturer's recommended diffusion coefficients and specifications (e.g., volume of resin; area and thickness of diffusion membrane) and the field-measured average water temperature.

4.2.2.2 Results of Metal Uptake Studies

The DGT data presented in Appendix K-1 and K-2 were reviewed in terms of evidence for bioavailability of the primary COPC (copper), and also provide additional evidence to confirm that other metals do not require designation as COPCs.

- **Copper**—In 2015, copper was detected in DGT samplers (µg/sampler) deployed in the Hazeltine Channel and Polley Lake, but was less than analytical detection limits in the samplers deployed in Bootjack Lake as well as both locations in Quesnel Lake. In 2016, copper was detected in all DGT samplers as a result of the improved analytical detection limits (Table 2 of Appendix K-2). Concentrations of copper in DGT samplers were approximately 3.5-times higher in Polley Lake than Bootjack Lake, approximately 2-times higher in Quesnel Lake impacted areas (i.e., near and farfield littoral and profundal locations) relative to Quesnel Lake reference and approximately 13-times higher in the Hazeltine Channel than Edney Creek. The copper concentrations were nevertheless low, despite the increase relative to reference. The DGT data indicate that exposed areas exceed reference areas with respect to copper exposure, and support the decision to select copper as the primary COPC. Back-calculated water concentrations for copper are discussed in further detail in Section 4.2.2.3.
- **Arsenic**—Arsenic was not identified as a COPC in surface water, but was identified as a COPC in sediment, and therefore is of interest to determine whether exposures near the sediment-water interface result in DGT uptake. Arsenic was not detected in any of the DGT samplers in 2015 (Table D.1 of Appendix K-1). In 2016, arsenic was detected in all DGT samplers (Table 2 of Appendix K-2). Average concentrations of arsenic were approximately 4-times higher in Polley Lake than Bootjack Lake. In Quesnel Lake, average concentrations of arsenic were higher in reference samples (0.024 µg/sampler) than impacted areas (0.014 µg/sampler) in profundal samples and similar between reference (0.018 µg/sampler) and impacted areas (0.017 µg/sampler) in littoral samples. Average concentrations of arsenic were similar between the Hazeltine Channel (0.095 µg/sampler) and Edney Creek (0.11 µg/sampler). Overall, the DGT samplers indicate that differences in arsenic exposure are minor and do not indicate a consistent increase in exposure in tailings exposed areas relative to reference.
- **Other Metals**—Other metals and metalloids were not identified as COPCs based on the available surface water data collected from the Hazeltine Channel (Section 2.4.2). An additional check was performed using the DGT information to confirm that these metals and metalloids are not present in a form and/or magnitude that would contribute to meaningful bioaccumulation (Table 1 of Appendix K-1). Overall, the DGT sampling reaffirms the appropriateness of emphasizing copper as the primary aqueous COPC, with no indications that other metals warrant consideration in the risk assessment as additional COPCs.

4.2.2.3 Implications Regarding Magnitude of Hazard

Copper was identified as a COPC in the problem formulation and was detected in the DGT samplers in the Hazeltine Channel and Polley Lake in both sampling years. The average “freely-available” (i.e., DGT-labile) water concentrations were back-calculated by the analytical laboratory based on the measured mass of copper in the DGT resin using the manufacturer’s recommended diffusion coefficients and specifications (e.g., volume of resin; area and thickness of diffusion membrane) and the field-measured average water temperature. The specific formula and the variables are provided in Appendix K-1. The resulting back-calculated DGT-labile water concentrations were compared to the site-specific total and dissolved copper concentrations for samples collected by Minnow before and after the deployment of the DGT devices.

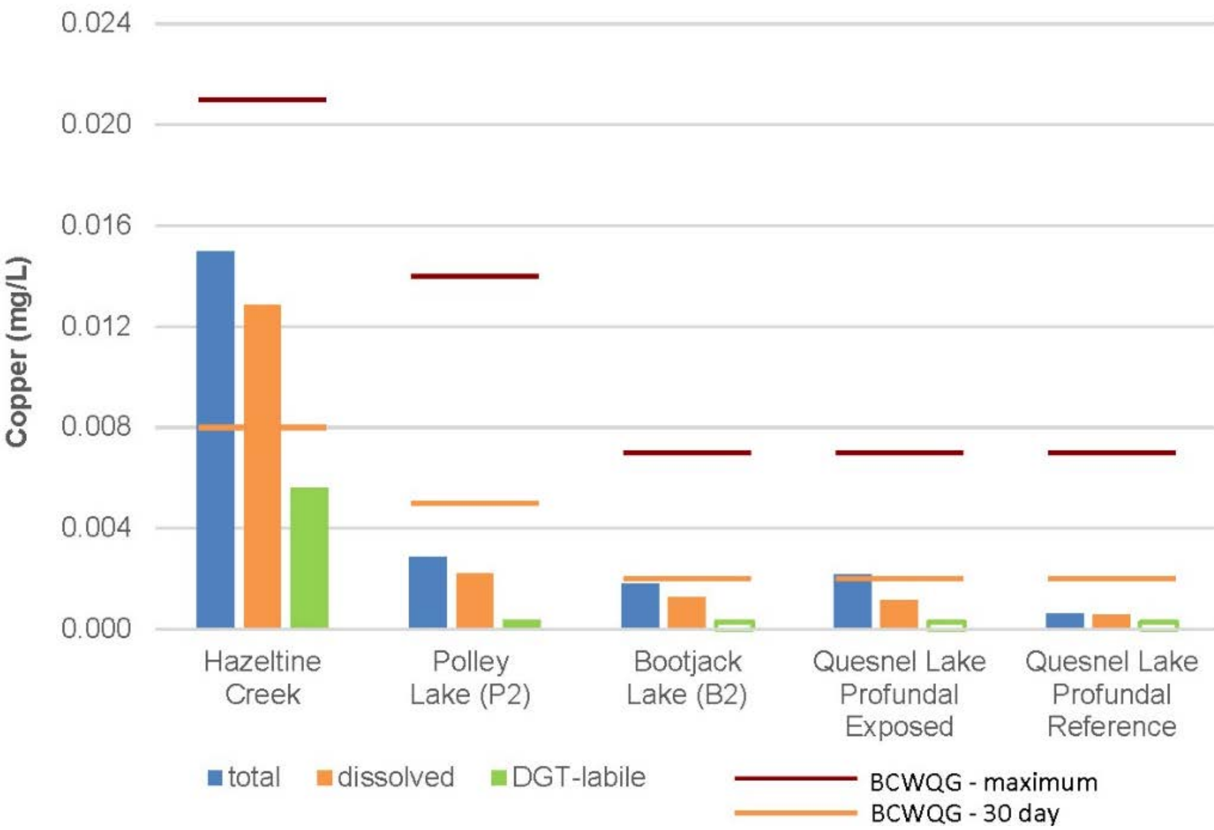

Figure 26: Average DGT-Available Copper Concentrations from 2015 Sampling Program (From Figure 2 of Appendix K-1)

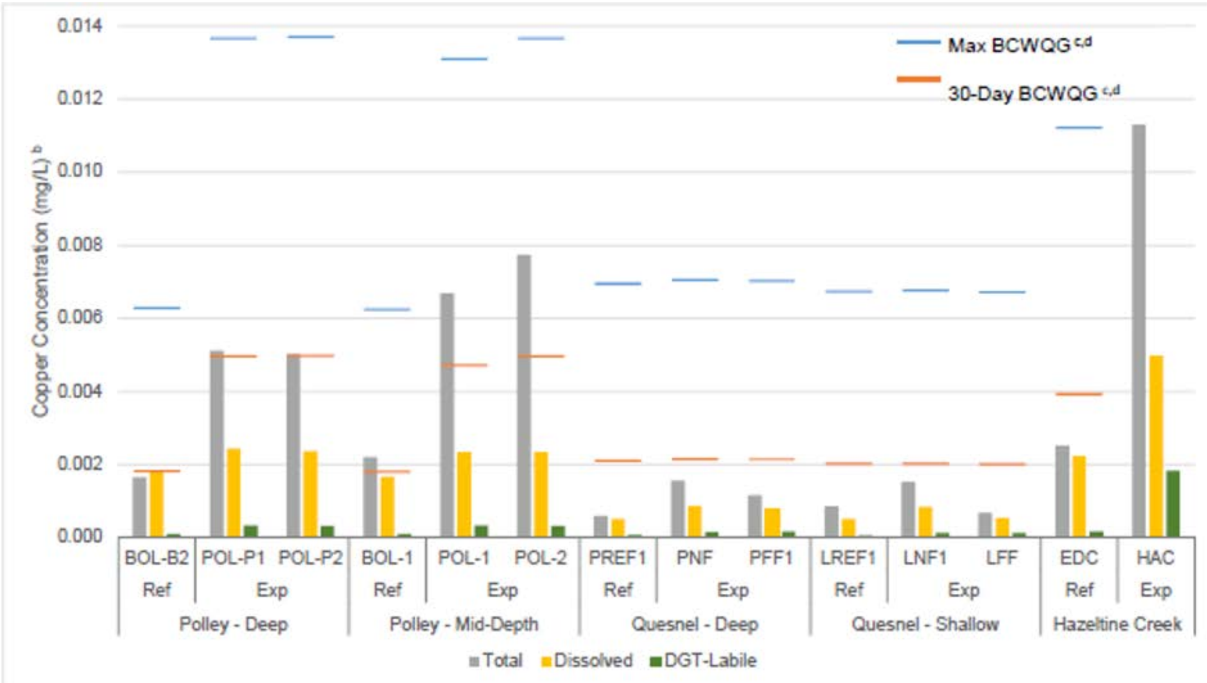

Figure 27: Average DGT-Available Copper Concentrations from 2016 Sampling Program (From Figure 2 of Appendix K-2)

Figure 26 and Figure 27 highlight that an exceedance of the ambient WQG for total copper does not necessarily mean that the copper is present in a form that would be readily available for uptake by aquatic organisms. Although ambient WQGs are generally intended to be applied to the total form of a metal, the dissolved concentration is known to be a more reliable predictor of toxicity and is the main form used in the studies used to derive those WQG. The copper WQG allows for the form of the copper compared to the WQG to be modified if detailed knowledge regarding copper bioavailability is available[33]. From a risk perspective, the use of DGT-labile, or freely-available, concentrations provides the necessary detailed knowledge—the DGT-labile concentrations provide a more realistic measure of the bioavailable fraction than either total or dissolved concentrations. Examples from the literature that are specific to copper in freshwater surface samples include:

- Tusseau-Vullemin et al. (2004) found the concentration-response curve and resulting EC_{50} from acute *Daphnia magna* toxicity testing was consistent between the total copper concentration in a spiked test using laboratory water (i.e., the basis for water quality guidelines) and the DGT "freely-available" fraction. The study was primarily focused on showing that DGT was measuring the bioavailable fraction in a manner consistent with the mechanistic understanding of copper bioavailability (e.g., the free-ion activity model and other aspects of biotic ligand modelling that are frequently used to predict the bioavailable fraction).

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[33] BC MoE (1987): Footnote 2 of Table 1 reads "when detailed knowledge on the bioavailable forms of copper is available, the form of copper in the criteria for aquatic life can be modified, as justified by the data"

- Apte et al. (2006) showed a linear relationship (R^2>0.95) between DGT-measured and total copper EC_{50} values for bacteria, algae, and cladocerans[34] exposed to copper spiked into natural waters with a wide range of dissolved organic carbon. DGT-measured concentrations were found to provide a conservative over-prediction of whether or not a water sample would demonstrate adverse effects that was still more realistic than relying on dissolved metals concentrations. This study showed that dose-response relationships for a variety of freshwater species were correlated between DGT and total concentrations in a predictable way, and that DGT was an effective approach for measuring the bioavailable fraction.
- Martin and Goldblatt (2007) found that 7-d *C. dubia* LC_{50} and EC_{50} values for field-collected water samples downgradient of a copper mine-impacted lake over the course of a year were more consistent (and lower) when expressed based on DGT-measured concentrations, but highly variable when based on the filterable copper concentrations[35]. This study shows that seasonal and other changes in the factors that influence copper bioavailability result in a variable and less realistic measurement of effects in a standardized toxicity test. Conversely, using DGT-measured concentrations reduced the variability in the measured effects because the DGT was providing a more realistic measure of the bioavailable fraction of copper in the water samples.
- Balistreri et al. (2007) used field-deployed DGT samplers as a line of evidence in the investigation by the US Geological Survey to determine the bioavailable fraction of cadmium, copper, nickel, and zinc in the waterbodies around the Elizabeth Copper Superfund site (Vermont, USA). DGT data were compared to literature LC_{50} values for fathead minnows and cladocerans as well as ambient water quality guidelines to determine which areas demonstrated an unacceptable hazard to aquatic organisms. This study provides an example of how DGT was incorporated by a regulatory agency into a contaminated site risk assessment.

Overall, the site-specific DGT data, in conjunction with the scientific literature shows that exceedance of the ambient WQG for total copper does not necessarily mean that the copper is present in a form that would be readily available for uptake by aquatic organisms. The DGT-labile data confirms that copper bioavailability is low and that the concentrations of freely available copper in water are less than the WQG.

4.2.2.4 Relative Contributions of Physical and Chemical Stressors

This particular line of evidence is focused on chemical stressors only. Physical stressors are not a factor that could confound the interpretation of these data because DGT is designed to measure the bioavailable fraction of copper (and other metals) in a sample irrespective of the presence of other substances (e.g., suspended sediment; organic carbon) or factors (e.g., hardness, pH, humic acids) that influence the fraction of total copper that is actually bioavailable to aquatic organisms. In contrast to total copper concentrations in surface, the DGT data assist in discriminating between physical and chemical factors because concentrations associated with the non-available particulate bound fractions would not be incorporated into the resin.

[34] Apte et al. (2006) looked at 12 different natural waters with a 4-h radiochemical bacterial growth test, a 48-h *Chlorella* sp. growth test and a 48-h *C. dubia* survival test.

[35] Range of DGT toxicity values was 20 – 30 µg/L for LC_{50}; 15 – 24 µg/L for reproduction EC_{25}. Range of filterable toxicity values was 96 – 203 µg/L for LC_{50}; 75 – 156 µg/L for reproduction EC_{25}.

4.2.2.5 Comparison to Decision Criteria and Uncertainty Analysis

A quantitative decision rule or criterion for how the back-calculated freely-available (i.e., DGT-labile) water concentrations would be used to contribute to the risk conclusion was not established. The DGT data was intended to provide valuable context for the dissolved and total surface water concentrations observed in conjunction with other lines of evidence. Although the DGT data were considered qualitatively, it was still appropriate to consider factors that reduce or increase the uncertainty in the DGT data evaluation:

- DGT does not have a specific regulatory-approved installation and analytical method. However, DGT sampling was identified as an acceptable line of evidence for determining the cause of toxicity in environmental effects monitoring programs for metals mines (Environment Canada 2012). The underlying science and application of passive samplers in general and DGT devices in particular was considered to be robust and has been summarized by reputable non-governmental organizations (Society of Environmental Toxicology and Chemistry (SETAC) 2012) in addition to individual authors. The availability of defensible methods reduces uncertainty in this type of data.

- The specific DGT sampling and analytical methodology was described by Minnow and Maxxam Analytics (see the technical reports in Appendix K). There were sufficient quality assurance/quality control measures incorporated in the design and execution of the work to identify the data as representative and suitable for use in the risk assessment. Key elements of the quality assurance/quality control program included:

  - DGT samplers were deployed in groups of three to five samplers (station replicates) in a relatively small area, and the data interpretation in the technical reports was based on the average DGT concentrations to incorporate the inter-device variability.

  - Analyses in 2016 were conducted using ultra-trace methods reducing the uncertainty associated with non-detected parameters. A refined comparison of copper and arsenic concentrations in DGT samplers between impacted and reference areas was possible using the ultra-trace data.

  - The sampling program included fabrication blanks, method spikes, field blanks, and trip blanks. There were no indications of data quality problems reported by the laboratory or the consultant. The trip blank had detectable concentrations of copper which could indicate a possible source of contamination from handling. Minnow concluded that the detected copper concentration in the field blank was not an issue because field blanks and reference samples showed no indication of copper. Contamination, if present, would bias the measured DGT concentrations of copper upward (not downward).

  - The formulae and technical specifications used by Maxxam to back-calculate the freely-available water concentration were described in the technical report and were adopted based on manufacturers' recommendations.

- The DGT passive sampling devices integrate the metal exposure over the duration of their deployment (i.e., they provide a time-weighted average) that is consistent with the objective of evaluating exposures over a chronic exposure period. Any individual DGT measurement has lower uncertainty relative to a co-located water sample in terms of understanding the bioavailable fraction of metals at that station. When a limited number of co-located samples are used to derive a "bioavailable fraction" that is then applied to a larger water data set, the specifics of how that ratio was calculated require consideration. In the current application, the majority of DGT samplers were placed in the water column approximately 1 m above the sediment surface. The resulting ratio reflects the geochemistry at that location and therefore should not be applied to other scenarios (e.g., sediment or soil porewaters; other lakes with differing hardness) without consideration of those factors.

4.2.2.6 *Summary of LOE Decision*

No specific decision criterion was established for this line of evidence. The general conclusions based on the available DGT data are that:

- Copper was the only constituent that had a higher uptake in the exposed-versus-reference comparisons. This finding confirmed the selection of copper as the primary COPC for assessment of aquatic health effects. For copper, the magnitude of the differences between exposure and reference conditions was less than an order of magnitude, with the exception of the Hazeltine Channel.
- A substantial fraction of the total copper measured in surface water samples is in a form that is less bioavailable to aquatic organisms via direct contact exposure pathways. This suggests that at a minimum, any comparisons of total water concentrations to water quality guidelines are providing a highly conservative evaluation of potential hazards to aquatic life. Concentrations of DGT-labile copper were substantially lower than water quality guidelines.
- The screening of constituents in Section 2.0 identified that the total concentrations of several metals and metalloids were elevated in turbid samples collected from the Hazeltine Channel. The DGT sampling confirmed that the available fraction of these substances was low.

In summary, a comparison of the bioavailable fractions of the aqueous exposure concentrations to guidelines suggests that the exposures of aquatic organisms to copper and other tailings-associated metals in surface waters are unlikely to elicit acute or chronic toxicity. The concentrations of DGT-labile copper are below the chronic water quality criterion for all locations investigated in 2015 and 2016. The total dissolved copper concentrations are also close to (i.e., less than factor of two exceedance) or below the guideline for all locations in both years.

4.3 Effects-Based LOE: Primary Producers

4.3.1 Aquatic Plant and Alga Toxicity Testing

Direct measurement of adverse effects to aquatic receptors using laboratory-based toxicity testing on field-collected surface water samples is a common approach in risk assessment. Laboratory-based toxicity testing is used to set ambient water quality guidelines and is integral to the provincial and federal regulatory programs for managing industrial effluent discharges. Standardized toxicity testing methods are available for a large number of surrogate species, including plants (the vascular plant *Lemna minor* and the alga *Pseudokirchneriella subcapitata*).

4.3.1.1 *Study Design and Methodology*

Aquatic plant toxicity testing (72-h *P. subcapitata* growth inhibition; 7-d *L. minor* growth inhabitation) data were available for one or more samples collected during the following sampling events:

- Sampling was conducted by Minnow between August 2014 through September 2014 for two locations in Polley Lake, two discharges from Polley Lake to upper Hazeltine Channel, one location in Quesnel Lake at the mouth of the Hazeltine Channel, and one location in Quesnel River[36] (Appendix L-1). The toxicity tests were conducted using unfiltered samples.

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[36] POL-2 and POL-6, HAD-1 and HAD-2 (pumped from Polley Lake to the Hazeltine Channel), QUL-66, and QUR-1.

- All toxicity testing was conducted by Nautilus Environmental (Burnaby, BC) following standard Environment Canada methods as described in the technical reports noted above.
- Standardized test methods and validation criteria, including QA/QC practices, were applied to laboratory toxicity tests to confirm data reliability and repeatability. Additionally, the representative test species chosen for testing (i.e., *P. subcapitata* and *L. minor*) are widely-used test species for which standardized culture and testing procedures exist.

Further testing with plant species was not conducted in 2015.

4.3.1.2 Evidence of Toxicity to Plants

There was no evidence of acute toxicity to plants in representative, unfiltered samples collected immediately after the initial release of tailings (Table 27). This toxicity test was discontinued in subsequent monitoring events.

Table 27: Summary of Plant Growth Inhibition Toxicity Tests

Location	Sample	Date	72-h *P. subcapitata* growth inhibition		7-d *L. minor* growth inhibition	
			IC_{25} (% v/v)	IC_{50} (% v/v)	IC_{25} (% v/v)	IC_{50} (% v/v)
Polley Lake	HAD-1 [a]	13 August 2014	>95.2	>95.2	n/a	n/a
	HAD-1[a]	20 August 2014	>95.2	>95.2	>97	>97
	POL-6-14m	16 September 2014	>95.2	>95.2	>97	>97
Quesnel Lake	QUL66-40m	21 August 2014	>95.2	>95.2	>97	>97
	QUL66-40m	28 August 2014	>95.2	>95.2	>97	>97
Quesnel River	QUR-1	22 August 2014	>95.2	>95.2	>97	>97

m = metres depth; n/a = not tested; $IC_{25/50}$ = 25th or 50th percentile inhibitory concentration.

(a) pumped from Polley Lake.

4.3.1.3 Comparison to Decision Criterion

The lack of an appropriately matched field reference (i.e., background conditions of water quality conditions, without influence of tailings-related constituents) reduces the ability to discern smaller response sizes. Accordingly, samples were compared to laboratory negative controls, which do not provide the same degree of discriminatory power relative to well-matched reference waters. Consequently, decision criteria for the plant toxicity line of evidence were established based on toxicological benchmarks used in standard acute and sub-lethal toxicity tests. Specifically, the 25th percentile inhibitory concentration (IC_{25}; % volume/volume) was applied. At these concentrations, growth is not inhibited relative to the control by more than 25%. A comparison of the toxicity testing results to decision criteria is provided in Table 26.

Table 28: Comparison of Toxicity Testing Results and Decision Criteria for the Cladoceran Toxicity Line of Evidence

Period	Test Type	Decision Criterion	Area	Number of Tests (n)	Number of Tests Failing Criterion	Effects Concentration (% v/v)
August 2014 – February 2015	72-h growth inhibition of *Pseudokirchneriella subcapitata*	IC_{25}	Polley Lake[a]	3	0	NA
			Quesnel Lake	2	0	NA
			Quesnel River	1	0	NA
	7-d growth inhibition of *Lemna minor*	IC_{25}	Polley Lake[a]	3	0	NA
			Quesnel Lake	2	0	NA
			Quesnel River	1	0	NA

n = number; v/v = volume per volume; h = hour; NA = not applicable; d = day; IC_{25} = 25[th] percentile inhibitory concentration.

(a) Includes discharge pumped from Polley Lake.

An IC_{25} concentration for <90% v/v was considered to indicate significant ecological response.

Overall, the strength of the plant toxicity testing line of evidence comes from its broad acceptance as an indicator of primary productivity. Toxicity testing allows for the direct measurement of potential effects from water quality to the survival, growth, and reproduction of test organisms. However, toxicity testing is conducted under laboratory conditions and with cultured test organisms. On balance, the strength of association to the assessment endpoint is considered moderate.

The decision criterion was were set based on IC_{25} concentrations. The decision criterion was not met if a toxicological response was observed in any round of testing for organisms exposed to waters with any level of sample dilution beyond 90% v/v (Table 26). The decision was refined based on the degree to which responses, where observed, were linked to physical factors.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Aquatic plant toxicity testing	Protect primary producers from reductions in survival, growth or reproduction as a result of direct contact with water impacted by deposited tailings.	Conduct 72-h chronic water algal growth tests on field collected water samples.	Performance is not inhibited by more than 25% relative to control samples.	Polley Lake	Met decision criterion
				Quesnel Lake	
				Quesnel River	
		Conduct 7-d water vascular plant growth toxicity tests on field collected water samples.	Performance is not inhibited by more than 25% relative to control samples.	Polley Lake	Met decision criterion
				Quesnel Lake	
				Quesnel River	

h = hour; % = percent; d = day.

4.3.2 Primary Productivity – Biomass as Chlorophyll *a*

To evaluate whether phytoplankton were protected from reductions in survival, growth, or reproduction as a result of direct contact with water (Assessment Endpoint 3; Section 2.4.7), biomass of primary producers (using chlorophyll *a* as a surrogate measure) was compared between the exposure area and reference areas of Quesnel Lake to determine if there were any changes due to the TSF-embankment breach.

4.3.2.1 Study Design and Methods

The assessment of primary productivity as chlorophyll *a* relied on data collected by MPMC following the TSF embankment breach in accordance with the MPMC Post TSF-Breach 2015 Monitoring Plan[37] and subsequent versions of the MPMC comprehensive environmental monitoring plan as described in Section 4.2.1. Chlorophyll *a* concentrations were assessed by comparing data between reference and exposed station in Quesnel Lake and between pre- and post-breach data in Quesnel and Polley Lakes.

4.3.2.2 Summary of Findings

Based on a visual assessment of the chlorophyll *a* time series, biomass in the West Basin was similar to that in the lake east of Cariboo Island immediately following the breach and in the following productive season (Figure 28). Chlorophyll *a* in September 2004 (Hume et al. 2005) was similar to that observed post-breach, and in September of 1985 to 1990 (Nidle et al. 1994) was higher than observed post-breach. The most recent pre-breach data for chlorophyll *a* measured in Polley Lake is from baseline studies conducted in 1995 and 1996, in which mean chlorophyll *a* concentrations were reported to range from 0.4 to 1.0 µg/L (Minnow 2014). In comparison, chlorophyll *a* measured in late 2014 was higher, and in 2015 was variable among sampling events, with some concentrations reported as within this historical range (July and September) and some higher or lower than this range (Figure 29). These findings indicate that copper concentrations in water are unlikely to have a significant impact on primary productivity.

[37] MPMC. 2015. Post TSF-Breach Monitoring Plan – 2015, Revision 1. 8 April 2015. Submitted to BC Ministry of Environment.

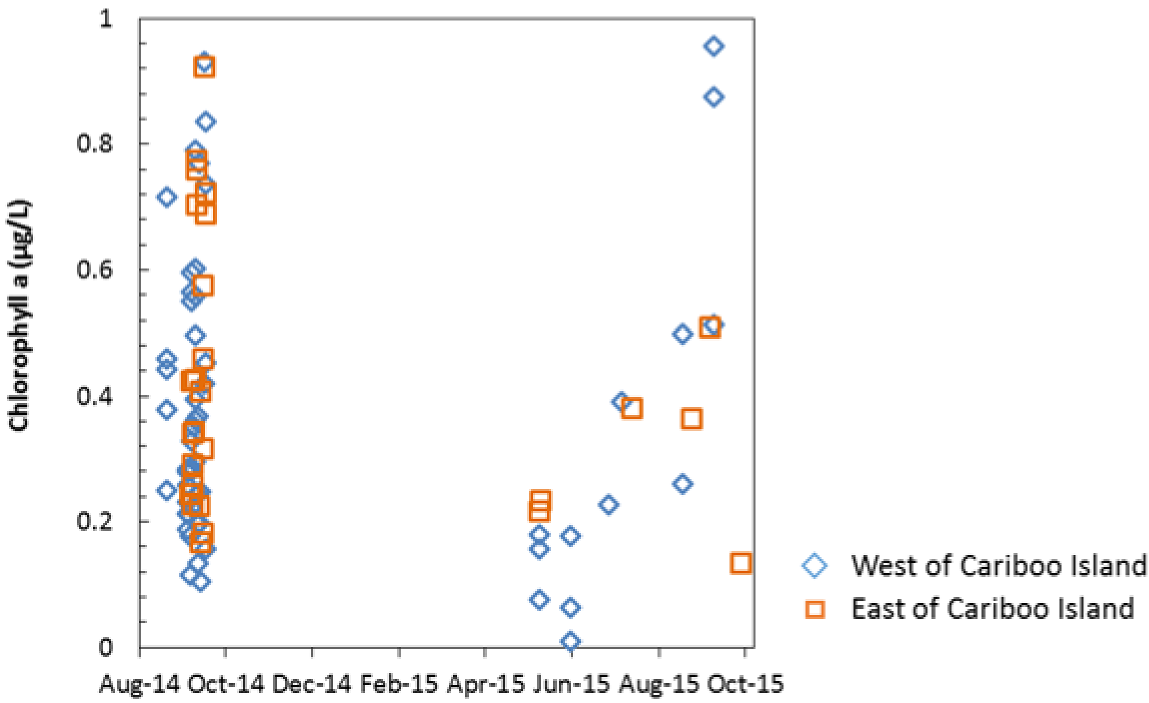

Figure 28: Temporal and Spatial Variability in Chlorophyll a *Concentration in Shallow (0 to 20 m) Water in the West Basin of Quesnel Lake (west of Cariboo Island; 11 stations) Compared to the Middle Arm of Quesnel Lake (east of Cariboo Island; 7 Stations)*

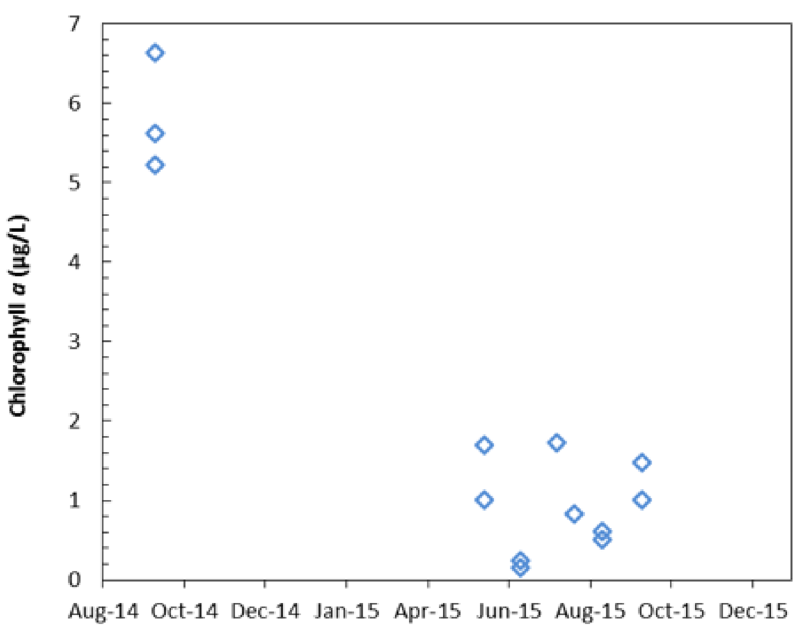

Figure 29: Temporal Variability in Chlorophyll a *Concentration in Polley Lake (2 stations)*

4.3.2.3 Confounding Factors

Plankton community metrics can be useful early indicators of environmental change because of their rapid response to changes in nutrients or other substances (e.g., Downing et al. 1990; Edmundson and Koenings 1986; Levine et al. 2005). However, the inherent variability within the plankton community (due to, for example, temperature and daylight) poses a challenge for interpreting changes including in the context of historical data, and also limits their usefulness as a monitoring tool. Plankton biomass varies vertically and horizontally within the open-water; therefore, estimates are sensitive to the number of stations, samples, and the depth of the water column sampled (Findlay and Kling 2001; Paterson 2002). Seasonal succession within the plankton community and natural year-to-year variation also contribute to the inherent variability of these communities (Wetzel 2001; Paterson 2002).

4.3.2.4 Comparison to Decision Criterion

The decision criterion for phytoplankton biomass was based on visual examination of plots showing biomass over time. There was no discernable difference in biomass in the exposed portion of Quesnel Lake relative to reference areas and chlorophyll *a* concentrations were within the range of historical values. In Polley Lake, chlorophyll *a* was notably higher in 2014 immediately following the TSF breach than measured historically which was more likely a response to nutrient inputs from the outwash materials rather than COPCs. Biomass was lower in 2015 and some samples were within the range of historical values.

The phytoplankton biomass line of evidence has a moderate level of uncertainty with respect to usefulness in making risk management decisions in response to the TSF breach:

- The line of evidence provides a direct community-level measurement of potential impacts on phytoplankton from contaminants in water; however, there is high natural variability in plankton communities, and although plankton communities were sampled at multiple times and locations during the open-water season, migration, aggregation, and predation can lead to patchy distributions that are difficult to characterize in field studies.
- Biomass of the exposed community in the West Basin falls within the range observed in the Middle Arm (reference)—although this does not provide a strong basis for discriminating potential minor alterations, it provides evidence that the West Basin conditions fall within a broad range of representative conditions for the region.
- Because the post-breach sampling in 2014 was conducted in the fall whereas the subsequent year (2015) sampling was conducted in the spring and/or summer, 2015 and 2014 data were not directly comparable.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Primary Productivity	Protect plankton from reductions in survival, growth or reproduction as a result of direct contact with water impacted by deposited tailings.	Measure the biomass of plankton in areas where tailings have been deposited.	Biomass has not changed relative to background areas, based on visual examination of plots.	Quesnel Lake	No apparent change
				Polley Lake	

4.4 Effects-Based LOE: Zooplankton

4.4.1 Cladoceran Toxicity Testing

Direct measurement of adverse effects to aquatic receptors using laboratory-based toxicity testing on field-collected surface water samples is a common approach in risk assessment. Laboratory-based toxicity testing is used to set ambient water quality guidelines and is integral to the provincial and federal regulatory programs for managing industrial effluent discharges. Standardized toxicity testing methods are available for a large number of surrogate species, including cladocerans (*Daphnia magna* and *Ceriodaphnia dubia*).

4.4.1.1 Study Design and Methodology

Cladoceran toxicity testing (48-h *D. magna* survival; 7-d *C. dubia* survival and growth) data was available for one or more samples collected during the following sampling events:

- Sampling was conducted by Minnow between August 2014 through September 2014 for two locations in Polley Lake, two discharges from Polley Lake to upper Hazeltine Channel, one location in Quesnel Lake at the mouth of the Hazeltine Channel, and one location in Quesnel River[38] (Appendix L-1).
- Sampling was conducted by Minnow between November 2014 and April 2015 for two locations in Polley Lake, one location in Quesnel Lake, and one location in Quesnel River. The Quesnel Lake and Quesnel River locations were the same as the initial sampling. The Polley Lake samples were collected from different locations[39] (Appendix L-2).
- Sampling was conducted by Minnow between June and November 2015 at one location in Polley Lake, one location in Quesnel Lake, and one location in Quesnel River. The locations were retained from the previous sampling design (with a minor change in location of the Quesnel Lake station)[40] (Appendix L-3).

All toxicity testing was conducted by Nautilus Environmental (Burnaby, BC) following standard Environment Canada methods as described in the technical reports noted above.

The initial toxicity tests for samples collected between August and September 2014 were conducted using unfiltered samples. Based on the responses observed in these *C. dubia* tests (Section 4.4.1.3), a subset of water samples collected between November 2014 and February 2015 were tested as separate unfiltered and filtered (0.45 µm) fractions. Testing of unfiltered and filtered samples was undertaken to determine the influence of suspended solids on toxicity responses (Appendix L-1 and L-2; MPMC 2015).

The follow-up testing completed between March and November 2015 was intended to validate the initial test results and focussed on the evaluation of unfiltered versus filtered samples, as well as the overall frequency and magnitude of the observed reproductive responses (Section 4.4.1.3). Similar to November 2014 through February 2015, survival and reproduction tests for *C. dubia* were completed using filtered (0.45 µm) and unfiltered water for a subset of samples to confirm the influence of suspended solids (Appendix L-3).

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[38] POL-2 and POL-6, HAD-1 and HAD-2 (pumped from Polley Lake to the Hazeltine Channel), QUL-66, and QUR-1.

[39] POL-P2 and POL-4, QUL-66, and QUR-1.

[40] POL-P2, QUL-55a (replaced QUL-66 in response to the change location of the Hazeltine Channel outflow), and QUR-1.

4.4.1.2 Evidence of Acute Toxicity to Cladocerans

There was no evidence of acute toxicity to cladocerans in representative, unfiltered samples collected immediately after the initial release of tailings (Table 29). This toxicity test was discontinued in subsequent monitoring events.

Table 29: Summary of 48-h *Daphnia magna* Survival Toxicity Tests

Location	Sample	Date	LC_{50} (% v/v)
Polley Lake	POL-2	9 August 2014	>100
	HAD-1[a]	13 August 2014	>100
	HAD-1[a]	20 August 2014	>100
	POL-6-14m	16 September 2014	>100
Quesnel Lake	QUL-66-40m	21 August 2014	>100
	QUL-66-40m	28 August 2014	>100
Quesnel River	QUR-1	6 August 2014	>100
	QUR-1	22 August 2014	>100

m = metres depth; LC_{50} = median lethal concentration.

(a) Pumped from Polley Lake.

4.4.1.3 Evidence of Chronic Toxicity to Cladocerans

There was no evidence of cladoceran mortality during longer-term (7-d) exposures in representative, unfiltered water samples collected from Polley Lake, Quesnel Lake, or Quesnel River. The toxicity tests also measured reproduction over the 7-d exposure period, and the concentration of site water that resulted in a 25% and 50% reduction in reproduction relative to the laboratory negative controls was reported. The sublethal reproduction endpoint is generally the most sensitive of the two test endpoints for this test protocol.

A summary of the available LC_{50}, IC_{25}, and IC_{50} values is provided in Table 30; the findings were:

- Chronic toxicity to cladocerans in Polley Lake was low, with no indications of toxicity that imply long-term or widespread potential for harm.
  - The majority of samples had no effects on survival or reproduction (i.e., the toxicity value is >100% v/v), with the exception of a single sample collected in September 2014 (POL-6-14m)
  - POL-6-14m was collected from a depth of 14 m, and was unremarkable in terms of its chemistry with the exception of a nitrite concentration of 0.22 mg/L (relative to the ambient WQG of 0.06 mg/L).
  - Minnow concluded that nitrite was unlikely to be a contributor to the toxicity because the measured concentration of 0.22 mg/L was less than a reproductive IC_{25} of 1.9 mg/L reported by US EPA (2010). Minnow concluded that elevated nitrite in POL-6-14m was likely associated with reducing conditions (consistent with the observed low dissolved oxygen levels in the field). The anomalously low dissolved oxygen concentrations were not observed in other samples collected by Minnow for toxicity testing from Polley Lake or Quesnel Lake, and no other samples had nitrite concentrations that exceeded the ambient water quality guideline.

  - The available information is not sufficient to definitively exclude nitrite or other water quality factors related to low dissolved oxygen in explaining the observed effect in POL-6-14m. It is not clear whether toxicity modifying factors such as hardness or chloride concentrations were consistent between Polley Lake and the testing conditions described by US EPA (2010), and as such the comparison to the reproductive IC_{25} of 1.9 mg/L is uncertain. Golder conducted a review of the bench sheets for POL-6-14m and found no issues in the test that could explain the findings. Nitrite may have been an anomalous contributing factor under the conditions present for this specific sample. Nitrite has not been an issue of toxicological concern in other samples collected.

- Chronic toxicity to cladocerans in Quesnel Lake and Quesnel River was present, but the results from side-by-side filtered and unfiltered samples indicate that effects were more likely the result of suspended particulates rather than dissolved metals or other constituents. Subsequent monitoring of suspended particulate in surface water samples indicates that these conditions are no longer present in these water bodies.

For water samples collected between August 2014 and September 2014, the toxicity test results indicated that toxicity was more likely attributed to turbidity, rather than elevated concentrations reported for some metals (Appendix L-1; MPMC 2015). Observed effects consisted primarily of impaired reproduction in *C. dubia* exposed to turbid, deep-water samples from Quesnel Lake.

No impacts on *C. dubia* survival were observed in water samples collected from Polley Lake, Quesnel Lake, or Quesnel River between November 2014 and February 2015. Similarly, no reproductive effects were observed in samples collected from Polley Lake and Quesnel River in January and February 2015 (Appendix L-2; MPMC 2015). Filtered samples from Quesnel Lake also failed to elicit reproductive effects in *C. dubia*; however, reproductive effects were observed in tests with unfiltered samples, particularly those collected at depth in Quesnel Lake near the Hazeltine Channel mouth and in the Quesnel River in November and December 2014 (Appendix L-2; MPMC 2015). Reproductive responses in the test samples from Quesnel Lake and Quesnel River appeared to be correlated with turbidity, rather than water chemistry. For example, the reproductive test response in unfiltered samples collected from the Quesnel River in November and December 2014 was coincident with rising turbidity levels at the sampling station (MPMC 2015).

Findings of the sampling and testing completed between March and November 2015 confirmed previous sampling results. No impacts to *C. dubia* survival were observed, and reproductive impacts were restricted to a sub-set of unfiltered samples collected from Quesnel Lake near the Hazeltine Channel mouth in January and March 2015, and one sample collected from the Quesnel River in March 2015 (Appendix L-3). Responses were attributed to turbidity, rather than elevated water chemistry, because concentrations of all analytes were below applicable BC WQGs and effects were not observed in sensitive fish species (Section 4.5.1) exposed to the same samples (Appendix L-3).

4.4.1.4 Relative Contribution of Physical and Chemical Stressors

Receiving environment waters were not acutely toxic to *D. magna*. Sub-lethal long-term effects to *C. dubia* were restricted to reproductive responses observed in a subset of unfiltered samples collected from Quesnel Lake near the Hazeltine Channel mouth in January and March 2015 (Appendix L-1, L-2, and L-3; MPMC 2015). These responses were considered to be attributed to turbidity in the unfiltered samples and not water chemistry, based on the following (Appendix L-3):

- Responses in paired filtered and unfiltered samples were restricted to the unfiltered samples
- Dissolved copper concentrations did not differ between paired filtered and unfiltered samples
- Effects were observed in unfiltered samples even when concentrations of water chemistry analytes (e.g., copper) were below applicable BC WQGs
- Observed responses in *C. dubia* did not align with fish toxicity testing results reported for the same samples, which showed little to no effects (Section 4.5.1)

Overall, the results indicate that responses were related to the elevated particulate matter present during turbidity events, not metals.

Table 30: Summary of 7-d *Ceriodaphnia dubia* Survival and Reproduction Toxicity Tests

Location	Sample	Date	LC_{50} (% v/v)	Reproduction IC_{25} (% v/v)	Reproduction IC_{50} (% v/v)
Polley Lake	POL-2	19 August 2014	>100	>100	>100
	HAD-1[a]	13 August 2014	>100	>100	>100
	HAD-1[a]	20 August 2014	>100	>100	>100
	HAD-1[a]	27 August 2014	>100	>100	>100
	HAD-2[a]	3 September 2014	>100	>100	>100
	HAD-1[a]	10 September 2014	>100	>100	>100
	POL-6-14m	16 September 2014	>100	3.8	5.3
	POL-4	16 December 2014	>100	>100	>100
	P2-0m	6 January 2015	>100	>100	>100
	P2-0m	14 April 2015	>100	>100	>100
	P2-0m	25 August 2015	>100	>100	>100
	P2-0m	11 November 2015	>100	>100	>100
Quesnel Lake	QUL-66-40m	21 August 2014	>100	3.9	9.8
	QUL-66-40m	28 August 2014	>100	3	5.3
	QUL-66-45m	3 September 2014	>100	1.8	13.6
	QUL-66-48m	10 September 2014	>100	<1.56	2.5
	QUL-66-40m	16 September 2014	>100	<1.56	7.3
	QUL-66-0m	25 November 2014	>100	29.3	>100
	QUL-66-0m (filtered)	25 November 2014	>100	>100	>100
	QUL-66-20m	25 November 2014	>100	6.2	>100
	QUL-66-20m (filtered)	25 November 2014	>100	>100	>100
	QUL-66-45m	25 November 2014	>100	22.1	>100
	QUL-66-45m (filtered)	25 November 2014	>100	92.5	>100
	QUL-66-0m	15 January 2015	>100	11.1	>100
	QUL-66-0m (filtered)	15 January 2015	>100	>100	>100
	QUL-66-85m	15 January 2015	>100	8.3	>100
	QUL-66-85m (filtered)	15 January 2015	>100	>100	>100
	QUL-66-0m	2 March 2015	>100	74.2	>100
	QUL-66-0m (filtered)	2 March 2015	>100	>100	>100
	QUL-55a-0m	25 August 2015	>100	>100	>100
Quesnel River	QUR-1	6 August 2014	>100	>100	>100
	QUR-1	22 August 2014	>100	>100	>100
	QUR-1	25 November 2014	>100	8.2	>100
	QUR-1	16 December 2014	>100	50.6	>100
	QUR-1	7 January 2015	>100	>100	>100
	QUR-1 (filtered)	7 January 2015	>100	>100	>100
	QUR-1	10 February 2015	>100	>100	>100
	QUR-1	3 March 2015	>100	95.9	>100
	QUR-1	24 August 2015	>100	>100	>100
	QUR-1	12 November 2015	>100	>100	>100

m = metres depth; LC_{50} = median lethal concentration; $IC_{25/50}$ = 25th or 50th percentile inhibitory concentration.

(a) Pumped from Polley Lake.

4.4.1.5 Uncertainty Analysis

Standardized test methods and validation criteria, including QA/QC practices, were applied to laboratory toxicity tests to confirm data reliability and repeatability. Additionally, the representative test species chosen for testing (i.e., *D. magna* and *C. dubia*) are sensitive, widely-used test species for which standardized husbandry and testing procedures exist. These species are also representative of the invertebrate order Cladocera, which is found in the management area for the Project.

Physical conditions, such as water clarity or turbidity, can confound determination of causal factors of toxicity (i.e., is the toxicity attributed to chemical or physical conditions?). To reduce uncertainty, toxicity tests with *C. dubia* were conducted using unfiltered and filtered water so that the influence of physical stressors (i.e., suspended solids) could be examined and therefore delineated from effects associated with water chemistry. Evaluating a range of samples that have both turbid and clear conditions, as well as lower and higher concentrations of dissolved metals helps to reduce this uncertainty.

Overall, the strength of the zooplankton toxicity testing line of evidence comes from its broad acceptance as an indicator of community health (particularly for the *C. dubia* reproduction endpoint), from the multiple rounds of testing that have been completed in different seasons and water quality conditions, and the paired assessment of filtered and non-filtered samples. Although there is uncertainty associated with the interpretation of individual test responses, the overall testing program provides confidence in the distinguishment of toxicity responses from natural variation, and for the identification of physical versus contaminant influences. Toxicity testing allows for the direct measurement of potential effects from water quality to the survival, growth, and reproduction of test organisms. However, toxicity testing is conducted under laboratory conditions and with cultured test organisms. On balance, the strength of association to the assessment endpoint is considered moderate.

4.4.1.6 Comparison to Decision Criteria

The decision criterion for the cladoceran toxicity line of evidence originally proposed by Golder was that toxicological performance is not reduced by more than 20% relative to background samples. However, the precision of this criterion was re-evaluated in consideration of the study design as implemented and variance observed within individual toxicity tests. The lack of an appropriately matched field reference (i.e., background conditions of water quality conditions, without influence of tailings-related constituents) reduces the ability to discern smaller response sizes. Accordingly, samples were compared to laboratory negative controls, which do not provide the same degree of discriminatory power relative to well-matched reference waters. Consequently, decision criteria for the cladoceran toxicity line of evidence were established based on toxicological benchmarks used in standard acute and sub-lethal toxicity tests. Specifically, the median lethal concentration (LC_{50}; % volume/volume) and the 25th percentile inhibitory concentration (IC_{25}; % volume/volume) were applied. At these concentrations, median survival under the current test conditions is not reduced by more than 50% and reproduction is not inhibited relative to the control by more than 25%. Any LC_{50} or IC_{25} concentrations less than or equal to 100% volume/volume were indicative of an effect and failure to meet decision criteria (i.e., decision criteria were not met if a toxicological response was observed in organisms exposed to diluted or undiluted water samples). Consideration of results from acute tests were also included with the chronic tests originally listed as the measurement endpoint. A comparison of the toxicity testing results to decision criteria is provided in Table 31.

Table 31: Comparison of Toxicity Testing Results and Decision Criteria for the Cladoceran Toxicity Line of Evidence

Period	Test Type	Decision Criterion	Area	Number of Tests (n)	Number of Tests Failing Criterion	Effects Concentration (% v/v)
August 2014 – February 2015	48-h acute lethality to *Daphnia magna*	LC_{50}	Polley Lake[a]	4	0	NA
			Quesnel Lake	2	0	NA
			Quesnel River	2	0	NA
	7-d survival and reproduction of *Ceriodaphnia dubia*	IC_{25}	Polley Lake[a]	9	1	3.8 to >100
			Quesnel Lake	10 unfiltered 5 filtered	10 unfiltered 0 filtered	<1.56 to 29.3 unfiltered 92.5 to >100 filtered
			Quesnel River	6 unfiltered 1 filtered	2 unfiltered 0 filtered	8.2 to >100
March – November 2015	7- to 8-d survival and reproduction of *Ceriodaphnia dubia*	IC_{25}	Polley Lake	3	0	NA
			Quesnel Lake	2 unfiltered 1 filtered	1 unfiltered 0 filtered	74.2 to >100
			Quesnel River	3	0	95.9 to >100

n = number; v/v = volume per volume; h = hour; LC_{50} = median lethal concentration; NA = not applicable; d = day; IC_{25} = 25th percentile inhibitory concentration; > = greater than; < = less than.

(a) Includes discharge pumped from Polley Lake.

An IC_{25} concentration for <90% v/v was considered t to indicate significant ecological response.

As described in Section 4.4.1.6, decision criteria were set based on LC_{50} or IC_{25} concentrations. Decision criteria were not met if a toxicological response was observed in any round of testing for organisms exposed to waters with a level of sample dilution beyond 90% v/v (Table 31). The decision criterion was refined based on the degree to which responses, where observed, were linked to physical factors.

LOE	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Cladoceran toxicity testing	Protect zooplankton from reductions in survival, growth or reproduction as a result of direct contact with water impacted by deposited tailings.	Conduct 48-h acute water flea survival tests on field collected water samples.	Median survival is not reduced by more than 50% relative to control samples.	Polley Lake	Met decision criterion
				Quesnel Lake	
				Quesnel River	
		Conduct 7-d water flea survival and reproduction toxicity tests on field collected water samples.	Performance is not inhibited by more than 25% relative to control samples.	Polley Lake	Met decision criterion
				Quesnel Lake	Failed to meet decision criterion under turbid conditions that no longer exist. Met decision criterion for non-turbid samples.
				Quesnel River	

LOE = line of evidence; h = hour; % = percent; d = day.

As summarized above, the only lethal or sublethal responses identified using the strict decision criterion for effects were reproduction inhibition in a subset of *C. dubia* tests of Quesnel Lake and Quesnel River waters. The criterion was met for all samples in all locations when results were filtered for applicability to representative conditions in these water bodies following stream rehabilitation.

4.4.2 Zooplankton Diversity, Abundance, and Bioaccumulation

To evaluate whether zooplankton were protected from reductions in survival, growth, or reproduction as a result of direct contact with water (Assessment Endpoint 3; Section 2.4.7), community metrics such as biomass, abundance, and taxonomic composition were compared between the exposure area and reference areas of Quesnel Lake to determine if there were any changes due to the TSF-embankment breach. Comparisons of tissue accumulations of copper in zooplankton were also made to evaluate relative exposures to this primary COPC.

4.4.2.1 Study Design and Methodology

Zooplankton samples for taxonomic analysis were collected between 2014 and 2016 from Quesnel Lake and between 2015 and 2016 from Polley Lake by MPMC staff following methods outlined in Appendix M-1 and summarized below:

- Samples were collected from three stations in Quesnel Lake: one exposed station called Hazeltine (QUL-Zoo-1; in the West Basin) and two reference stations, Horsefly (QUL-Zoo-7; near the Horsefly River) and Junction (QUL-Zoo-8; in the Main Basin where the East and North arms meet).
- Samples were collected from two stations in Polley Lake: P1 and P2.
- Sampling methods and taxonomic laboratories differed between Quesnel Lake and Polley Lake to be consistent with previous data collection methods and thereby allow comparison with previous years' data sets.

Data from the zooplankton monitoring sampling events are available in Appendix M-1 (2014 and 2015 data) and Appendix M-3 (2016 data). The data were evaluated as follows:

- Spatial and temporal trends in total abundance and biomass of major zooplankton taxonomic groups were qualitatively examined by plotting data.
- Relative abundance and biomass of major zooplankton groups were plotted by year for each sampling event.
- Due to differences in resolution of taxonomic identification as well as units [41], the zooplankton abundance and biomass data were not compared to pre-breach data presented in Hume et al. (2005) and MacLellan et al. (1993).

[41] For example, areal [value/m^2] versus volumetric [value/m^3] units and presentation on a dry- versus wet-weight basis used for the post-breach samples versus pre-breach sampling. Insufficient information regarding methods was available to convert data to a common unit.

There are no pre-breach or reference data against which to compare the Polley Lake community data, and zooplankton data from Quesnel and Polley Lakes are not comparable. Therefore Polley Lake was not assessed for this line of evidence.

4.4.2.2 Summary of Biological Metrics

The following observations of the zooplankton taxonomy data informed the conclusions for this receptor group:

- Total zooplankton biomass (Figure 30) and abundance (Figure 31) in Quesnel Lake were generally lowest at Horsefly and highest at Junction both during the fall of 2014 and through the spring/summer of 2015. In 2016, biomass and abundance were highest at Junction and lowest at Hazeltine.
- Zooplankton biomass was dominated by either cyclopoid copepods or cladocerans (Figure 32). Cyclopoid copepods were generally dominant in spring and early-summer and cladocerans dominant in late-summer and fall. Calanoid copepods were generally present in lower numbers compared to other taxa with the exception of May 2015 where approximately 51% of the zooplankton biomass at Horsefly and 44% at Hazeltine was composed of calanoid copepods.
- In terms of abundance, cyclopoid copepods were the dominant taxon at all stations (Figure 33). In 2015 and to a lesser extent in 2016, zooplankton abundance was composed primarily of cyclopoid and calanoid copepods earlier in the productive season followed by a shift to cladocerans representing the dominant taxa later in the season.

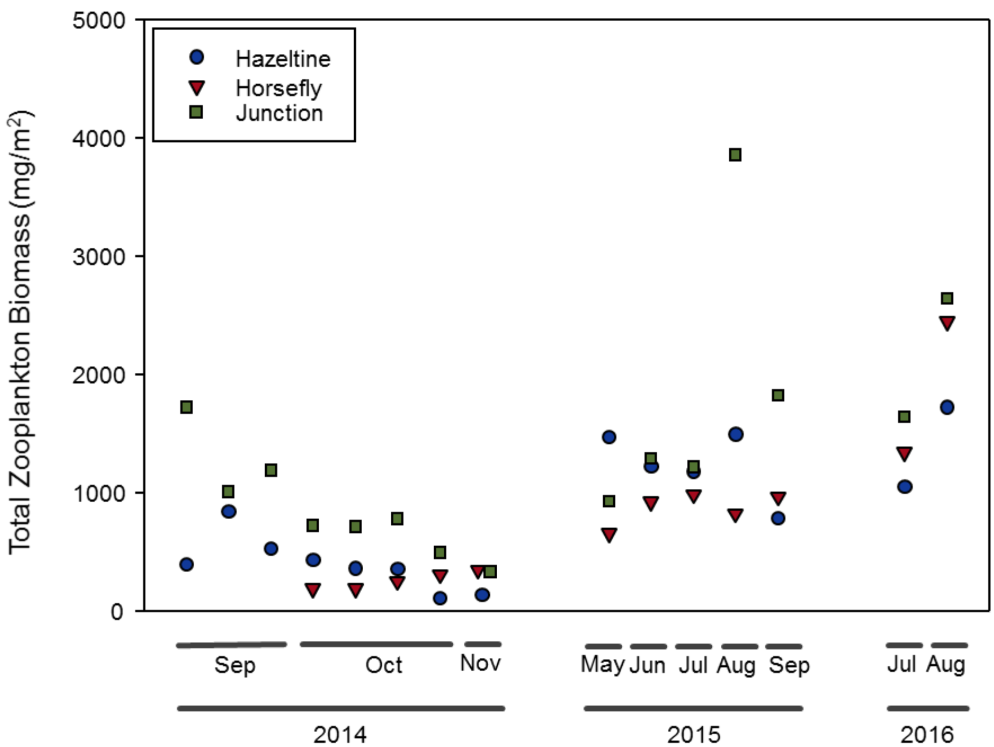

Figure 30: Total Zooplankton Biomass in Quesnel Lake, 2014 to 2016

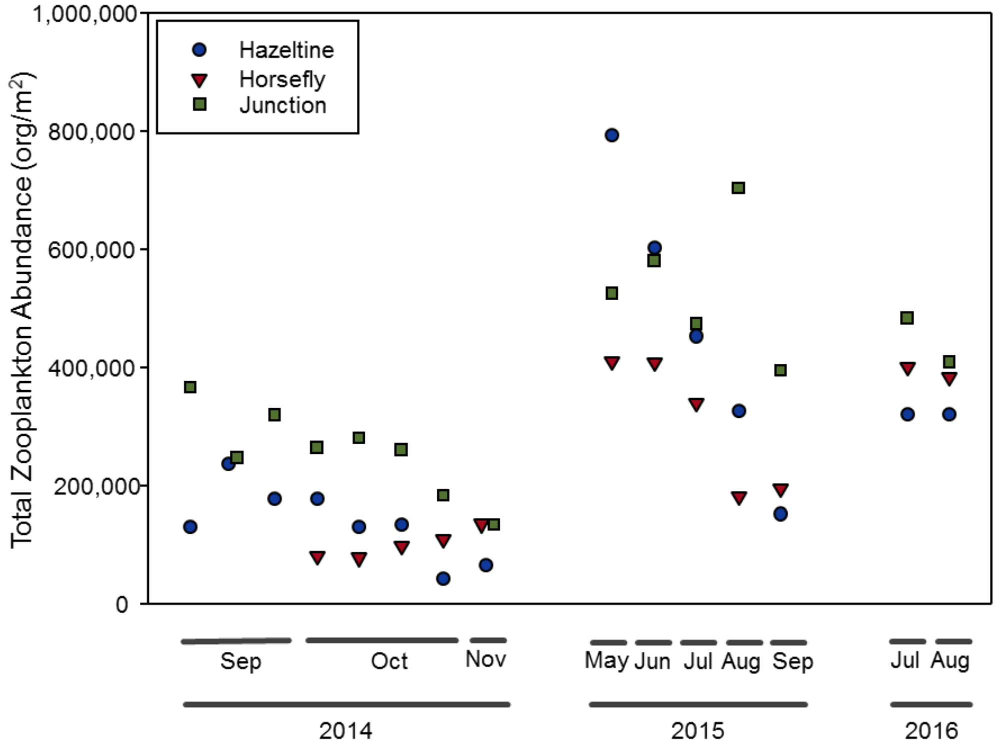

Figure 31: Total Zooplankton Abundance in Quesnel Lake, 2014 to 2016

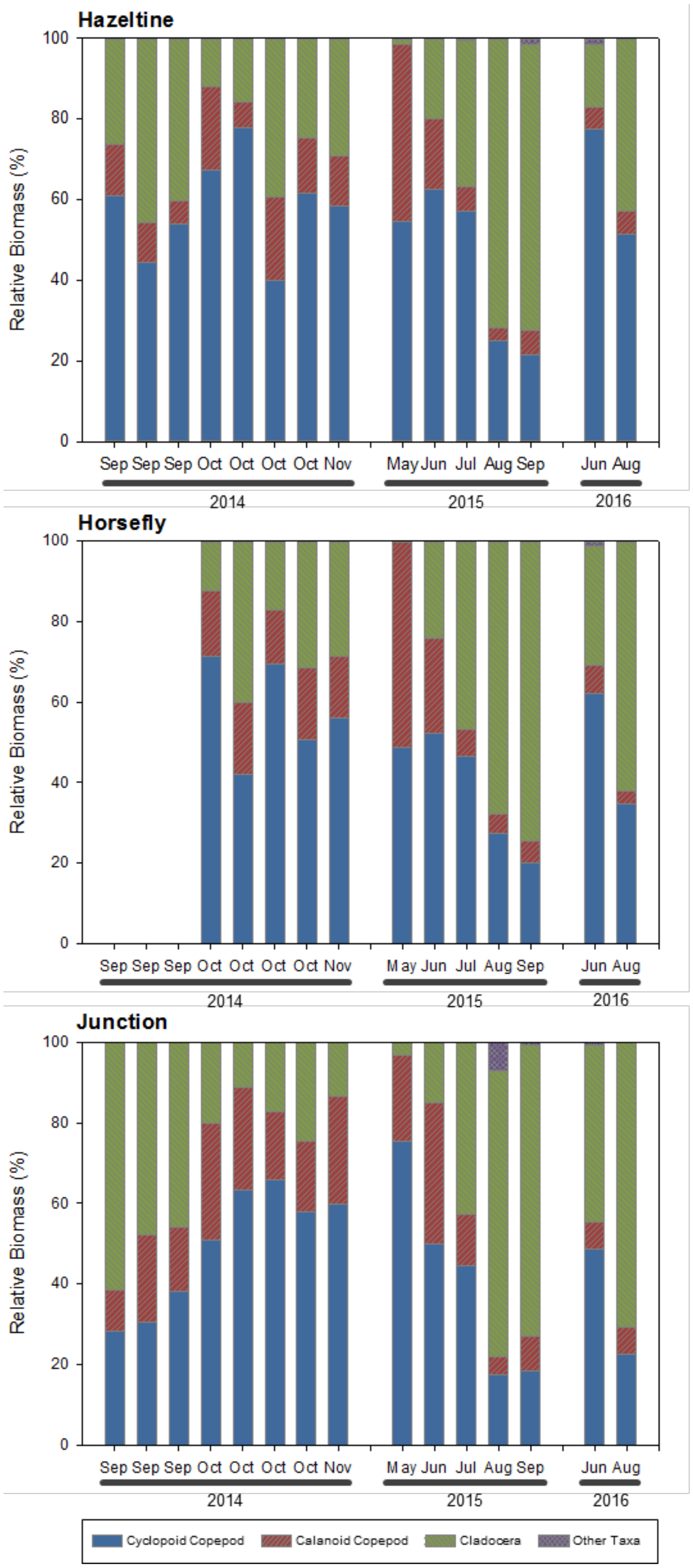

Figure 32: Composition of Zooplankton Biomass by Subtype at Quesnel Lake Monitoring Stations, 2014 to 2016

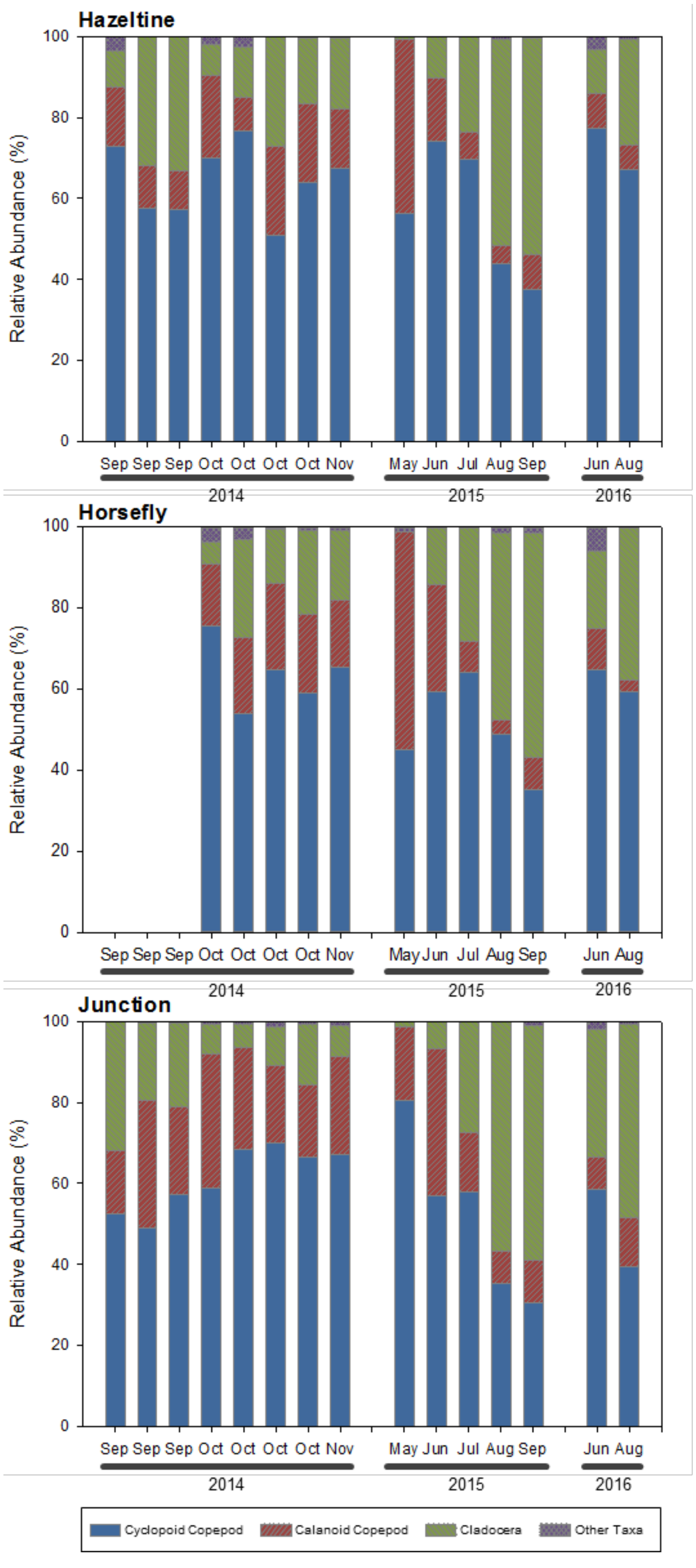

Figure 33: Composition of Zooplankton Abundance by Subtype at Quesnel Lake Monitoring Stations, 2014 to 2016

4.4.2.3 Summary of Copper Bioaccumulation

The detailed evaluation of copper bioaccumulation data in lower trophic level organisms (including zooplankton) is summarized in Section 4.5.2, in the context of potential differences in exposure to fish through consumption of prey. This section provides an overview of the findings for zooplankton bioaccumulation in the context of direct effects to the health of the zooplankton. The magnitude of potential effects due to the TSF embankment breach was determined by comparing copper concentrations in zooplankton tissue collected from exposed areas and reference areas. The concentrations of copper in zooplankton collected from reference areas of Quesnel Lake are considered representative of the natural range of variability, referred to as the normal range. Data from the two reference stations in Quesnel Lake (Horsefly and Junction) and all years of post-breach data (i.e., 2014 to 2016) were used to develop the natural range. Reference data were not available for Polley Lake and therefore the normal range of the Quesnel Lake reference data were used for comparison to Polley Lake exposed data. Details of the statistical processing and application of the decision criteria for bioaccumulation-based measures are summarized in Section 4.5.2. Overall, there was no discernable difference in copper concentrations of zooplankton tissue samples between the exposed area and the reference areas of Quesnel Lake or Polley Lake, which resulted in “no change” for this line of evidence in both water bodies.

4.4.2.4 Confounding Factors

Tissue bioaccumulation measurements provide estimates of absolute and relative exposure to zooplankton, but are confounded by aspects of sample collection and processing. The zooplankton community is a complex assemblage of diverse organism types (with phylogenetic differences in uptake and internal regulation of copper as an essential nutrient), and it is difficult to standardize the types of organisms collected among stations. In addition, the chemical measurements are subject to uncertainties related to low sample volumes and other analytical quantitation constraints. For this program, the relative percent difference (RPD) for copper between duplicate samples in the 2016 laboratory analyses exceeded the laboratory data quality objective of 40%, which was inferred to be attributable to sample heterogeneity.

In recognition of the above sources of variance, it was not considered possible to differentiate between physical factors and chemical factors as potential modifiers of plankton community composition and abundance. Furthermore the ability to detect meaningful (or statistically significant) alterations in abundance or composition is limited by the wide variance in natural conditions for these parameters.

4.4.2.5 Comparison to Decision Criteria

The primary decision criterion for zooplankton community metrics was based on visual examination of plots showing total and relative abundance and biomass over time.

- Total zooplankton biomass and abundance at Hazeltine were generally lower than observed at Junction for a given sampling period, but similar to or greater than values observed at Horsefly (Figure 30 and Figure 31). Because the overall biomass and abundance at Hazeltine was intermediate between the conditions observed at the other two stations, the breach did not appear to affect total zooplankton biomass or abundance at the exposed area of Quesnel Lake.

- Zooplankton biomass and abundance in Quesnel Lake were generally dominated by either cyclopoid copepods or cladocerans (Figure 32 and Figure 33). Similar seasonal trends observed at the Hazeltine, Horsefly, and Junction stations in each year suggest that the observed changes were regional rather than related to the breach.

There was no discernable response in total zooplankton biomass or abundance, or in the relative biomass or abundance of dominant taxa, which resulted in a “no change” for this line of evidence.

A secondary decision criterion for zooplankton community health was based on the observed bioaccumulation of copper in zooplankton tissue. To evaluate the magnitude of potential effects due to the TSF embankment breach, copper concentrations in zooplankton tissue collected from exposed areas were compared to reference areas. A non-parametric method was used to calculate the normal range which used the 2.5^{th} and 97.5^{th} percentiles as the lower and upper limits (Barrett et al. 2015). The median concentration of copper from the exposed area of Quesnel Lake (i.e., Hazeltine) and Polley Lake (P1 and P2 combined) was compared to the above reference normal range. There were no discernable differences in the copper bioaccumulation characteristics of exposed versus reference stations, which resulted in a “no difference” determination for this line of evidence in both Polley Lake and Quesnel Lake (Section 4.5.2).

The zooplankton community line of evidence has a moderate level of uncertainty with respect to usefulness in making risk management decisions in response to the TSF breach:

- The line of evidence provides a direct community-level measurement of potential impacts on zooplankton from contaminants in water; however, there is high natural variability in plankton communities, and although plankton communities were sampled at multiple times and locations during the open-water season, migration, aggregation, and predation can lead to patchy distributions that are difficult to characterize in field studies.
- The community abundance and composition of the exposed community at Hazeltine falls within the envelope of conditions bounded by the two far-field locations (Horsefly and Junction)—although this does not provide a strong basis for discriminating potential minor alterations, it provides evidence that the Hazeltine conditions fall within a broad range of representative conditions for the region.
- Because the post-breach sampling in 2014 was conducted in the fall whereas subsequent years’ (2015 and 2016) sampling was conducted in the spring and/or summer, 2015/2016 data and 2014 data were not directly comparable. In addition, due to differences in resolution of taxonomic identification as well as units, zooplankton abundance and biomass data cannot be compared to pre-breach data. The magnitude of the short-term impact of the original tailings deposition may mask an underlying influence related to a stressor that will only become evident based on longer term monitoring.

Comparison of tissue chemistry of zooplankton in the management area to reference areas (i.e., background conditions) provides context for potential breach-related impacts, but is a less reliable indicator of response related to effects-based measures. Changes in tissue chemistry of prey items relative to background is a measurement of exposure and does not necessarily mean that an adverse effect is occurring at the individual, population, or community level.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Zooplankton community	Protect zooplankton from reductions in survival, growth or reproduction as a result of direct contact with water impacted by deposited tailings.	Measure the diversity, biomass, and abundance of zooplankton in areas where tailings have been deposited.	Biomass or abundance has not changed relative to background areas, based on visual examination of plots.	Quesnel Lake	No apparent change
Zooplankton tissue chemistry	Protect invertebrates from accumulations of tissue-bound copper above reference levels.	Measure the concentration of copper in field-collected samples of zooplankton.	The median concentration in samples collected from the management area is lower than the upper limit of the normal range of samples collected from reference areas.	Quesnel Lake	No change
				Polley Lake	No change

4.5 Effects-Based LOE: Fish

4.5.1 Larval Fish Toxicity Testing

4.5.1.1 Study Design and Methodology

Fish toxicity testing data (96-h rainbow trout survival; 7-d survival and growth of juvenile fathead minnows and rainbow trout, survival and development of embryo-alevin stages of rainbow trout) were available for one or more samples collected during the following sampling events:

- Sampling was conducted by Minnow between August 2014 through September 2014 at two locations in Polley Lake, two discharges from Polley Lake to upper Hazeltine Channel, one location in Quesnel Lake at the mouth of the Hazeltine Channel, and one location in Quesnel River[42] (Appendix L-1).
- Sampling was conducted by Minnow between November 2014 and April 2015 at one location in Polley Lake, one location in Quesnel Lake, and one location in Quesnel River. The Quesnel Lake and Quesnel River locations were the same as those from the initial sampling. The Polley Lake samples were collected from different locations[43] (Appendix L-2).
- Sampling was conducted by Minnow between June and November 2015 at one location in Polley Lake, two locations in Quesnel Lake, and one location in Quesnel River. The locations were retained from the previous sampling (with a minor change in location of the Quesnel Lake station)[44] (Appendix L-3).

[42] POL-2 and POL-6, HAD-1 and HAD-2 (pumped from Polley Lake to the Hazeltine Channel), QUL-66, and QUR-1.

[43] POL-P2, QUL-66, and QUR-1.

[44] POL-P2, QUL-55 and then QUL-55a (replaced QUL-66 in response to the change location of the Hazeltine Channel outflow) and, QUR-1.

Toxicity testing was conducted by Nautilus Environmental (Burnaby, BC) following standard methods (i.e., Environment Canada and others) as described in the technical reports noted above.

Based on unexpected sublethal growth responses in the initial 7-d fathead minnow tests for samples collected between August 2014 and September 2014 (Appendix L-1; Section 4.5.1.3), additional 7-d tests (i.e., follow-up confirmatory tests) were run. These tests included parallel testing of fathead minnows and rainbow trout to confirm or refute the potential for growth effects in fish.

In addition to the standard 96-h acute lethality and 7-d survival and growth tests that were carried out on rainbow trout and fathead minnow, two early life stage survival and development tests were completed on rainbow trout using embryo and alevin life stages (Appendix L-2). The embryo-alevin tests of rainbow trout have a longer exposure duration (approximately one month), incorporate lethal and sublethal test endpoints including developmental markers, and generally are more sensitive to the identification of chemical-induced responses relative to the standard 96-h acute lethality test. Water samples collected regularly from the Quesnel River between 25 November 2014 and 7 January 2015 were used in the tests; the sample collection period was scheduled to coincide with egg availability and the period of greatest turbidity in Quesnel River.

4.5.1.2 Acute Toxicity to Fish

There was no evidence of acute toxicity to rainbow trout in representative samples collected immediately after the initial release of tailings. This toxicity testing was discontinued in subsequent monitoring events.

Table 32: Summary of 96-h Rainbow Trout Survival Toxicity Tests

Location	Sample	Date	LC_{50} (% v/v)
Polley Lake	POL-2	9 August 2014	>100
	HAD-1[a]	13 August 2014	>100
	HAD-1[a]	20 August 2014	>100
	POL-6-12m	30 September 2014	>100
Quesnel Lake	QUL-66-40m	21 August 2014	>100
	QUL-66-40m	28 August 2014	>100
Quesnel River	QUR-1	6 August 2014	>100
	QUR-1	22 August 2014	>100

m = metres depth; LC_{50} = median lethal concentration.

(a) Pumped from Polley Lake.

4.5.1.3 Chronic Toxicity to Fish

There was no evidence of fish mortality during longer-term (7-d or 31-d) exposures in representative water samples collected from Polley Lake, Quesnel Lake, or Quesnel River, with the exception of one sample from the outlet of Polley Lake. The toxicity tests also measured growth over the 7-d exposure period in both fathead minnow and rainbow trout tests, and evaluated normal hatching and development of rainbow trout embryos over a 31-d exposure period. The concentrations of site water that resulted in a 25% and 50% reduction in these endpoints relative to the laboratory negative controls were reported. A summary of the available LC_{50}, IC_{25}, and IC_{50} values is provided in Table 33, and indicates that:

- No exposure-related impacts on survival and growth of rainbow trout or fathead minnows were observed for water samples collected between August 2014 and November 2015. There were effects in a small number of samples collected between August and September 2014, but these were not correlated with metal concentrations. (Appendix L-1 and L-2; MPMC 2015). No impacts on survival or growth of rainbow trout or fathead minnow were observed in follow-up confirmatory testing of water samples collected from Polley Lake, Quesnel Lake, or Quesnel River (Appendix L-3).
- No adverse effects on the survival or development of rainbow trout eggs or alevin were observed during the 31-d toxicity tests completed using water collected from the Quesnel River between November 2014 and January 2015, a period of elevated turbidity. Consequently, it was concluded that the increase in turbidity in the Quesnel River after turnover of Quesnel Lake was unlikely to affect salmonid eggs incubating in the river (Appendix L-2; MPMC 2015).

Table 33: Summary of Chronic Survival and Growth Toxicity Tests with Fathead Minnow and Rainbow Trout

Location	Test/Species	Sample	Date	LC_{50} (% v/v)	Growth[c] IC_{25} (% v/v)	Growth[c] IC_{50} (% v/v)
Polley Lake	7-d fathead minnow	HAD-1[a]	13 August 2014	>100	>100	>100
		HAD-1[a]	20 August 2014	>100	>100	>100
		HAD-1[a]	27 August 2014	>100	71.6	>100
		HAD-2[a]	3 September 2014	84.1	24.2	>100
		HAD-1[a]	10 September 2014	>100	>100	>100
		POL-6-14m	16 September 2014	>100	>100	>100
		P2-0m	6 January 2015	>100	>100	>100
		P2-0m	14 April 2015	>100	>100	>100
	7-d rainbow trout	P2-0m	6 January 2015	>100	>100	>100
		P2-0m	14 April 2015	>100	>100	>100
		P2-0m	25 August 2015	>100	>100	>100
		P2-0m	11 November 2015	>100	>100	>100
Quesnel Lake	7-d fathead minnow	QUL-66-40m	21 August 2014	>100	>100	>100
		QUL-66-40m	28 August 2014	>100	>100	>100
		QUL-66-45m	3 September 2014	>100	>100	>100
		QUL-66-48m	10 September 2014	>100	>100	>100
		QUL-66-40m	16 September 2014	>100	>100	>100
		QUL-66-0m	15 January 2015	>100	83.2[b]	>100
		QUL-66-85m	15 January 2015	>100	95.6[b]	>100
		QUL-66-0m	2 March 2015	>100	>100	>100
	7-d rainbow trout	QUL-66-0m	15 January 2015	>100	>100	>100
		QUL-66-85m	15 January 2015	>100	>100	>100
		QUL-55-0m	16 June 2015	>100	>100	>100
		QUL-55a-0m	25 August 2015	>100	>100	>100

Location	Test/Species	Sample	Date	LC_{50} (% v/v)	Growth[c] IC_{25} (% v/v)	Growth[c] IC_{50} (% v/v)
Quesnel River	7-d fathead minnow	QUR-1	22 August 2014	>100	75.9	>100
		QUR-1	7 January 2015	>100	>100	>100
		QUR-1	10 February 2015	>100	>100	>100
		QUR-1	3 March 2015	>100	>100	>100
	7-d rainbow trout	QUR-1	7 January 2015	>100	>100	>100
		QUR-1	16 June 2015	>100	>100	>100
		QUR-1	24 August 2015	>100	>100	>100
		QUR-1	12 November 2015	>100	>100	>100
	31-d rainbow trout embryo/alevin	QUR-1	25 November 2014	>100	>100	>100
		QUR-1	9 December 2014	>100	>100	>100

m = metres depth; LC_{50} = median lethal concentration; $IC_{25/50}$ = 25th or 50th percentile inhibitory concentration.

(a) pumped from Polley Lake.

(b) IC25 and IC50 for growth when expressed on a dry weight basis were >100% v/v; therefore, growth was not considered to be affected in these tests.

(c) ICx for growth expressed on wet weight basis; ICX is for development in 31-d embryo/alevin test.

4.5.1.4 Confounding Factors

Exposure to water samples collected between August 2014 and November 2015 resulted in no adverse impacts to survival and growth of rainbow trout and fathead minnow (Appendix L-1, l-2, L-3). Although some anomalous responses were observed in Fall 2014 for the 7-d fathead minnow test, these responses were not observed in subsequent validation tests with trout and minnows that included a longer-duration (and generally more sensitive) chronic development test with rainbow trout. Therefore, the relative contribution of physical and chemical stressors was not assessed.

4.5.1.5 Uncertainty Analysis

Standardized test methods and validation criteria, including QA/QC practices, were applied to laboratory toxicity tests to confirm data reliability and repeatability. Additionally, the selected test species (i.e., rainbow trout and fathead minnow) are sensitive, widely-used test species for which standardized husbandry and testing procedures are available, including standard Environment Canada protocols. Rainbow trout are also present within the management area and fathead minnows are considered representative of small bodied fish (cyprinid family of fishes) also found within the management area.

It is possible that laboratory-reared test organisms may be more sensitive to contaminants or contaminant mixtures than organisms that have acclimated to site conditions and developed tolerances to concentrations of metals that may impact laboratory-reared organisms. Toxicity tests completed in the laboratory may not fully capture all the natural variables that influence bioavailability and toxicological responses of fish. Laboratory tests are also not indicative of how the broader aquatic community responds to the substances or mixtures being tested.

Toxicity testing allows for direct measurement of potential effects from water quality to the survival, growth, and reproduction of test organisms. However, toxicity testing is completed under laboratory conditions and with cultured test organisms. Therefore, the strength of association to the assessment is considered moderate.

A specific uncertainty for this testing program relates to the interpretation of the anomalous toxicity results for fathead minnows tested in the late fall of 2014. These responses do not appear to be related to water chemistry, and confirmatory testing in subsequent seasons did not identify toxicity to early life stages of either minnows or trout, even when the test protocol was extended in duration for rainbow trout. Furthermore, these samples consisted of pumped water samples from Polley Lake that no longer reflect site-specific conditions.

4.5.1.6 Comparison to Decision Criteria

The default decision criterion for the larval fish toxicity line of evidence was that toxicological performance is not reduced by more than 20% relative to background samples. This criterion was switched to the 25th percentile inhibitory concentration (IC_{25}; % v/v) based on comparison to negative controls to correspond to the standard metrics calculated by the toxicology laboratory. This is more conservative than the default criterion applied. A comparison of the toxicity testing results to decision criteria is provided in Table 34.

Table 34: Comparison of Toxicity Testing Results and Decision Criteria for the Larval Fish Toxicity Line of Evidence

Period	Test Type	Decision Criterion	Area	Number of Tests (n)	Number of Tests Failing Criterion	Effects Concentration (% v/v)
August 2014 – February 2015	96-h acute lethality to rainbow trout	LC_{50}	Polley Lake[a]	4	0	>100
			Quesnel Lake	2	0	
			Quesnel River	2	0	
	7-d survival and growth of fathead minnow	IC_{25}	Polley Lake[a]	7	2	24.2–100 (but not related to water chemistry)
			Quesnel Lake	7	0	
			Quesnel River	3	1	
	7-d survival and growth of rainbow trout	IC_{25}	Polley Lake	1	0	>100
			Quesnel Lake	2	0	
			Quesnel River	1	0	
	31-d survival and development of rainbow trout	IC_{25}	Quesnel River	2	0	>100
March – November 2015	7-d survival and growth of fathead minnow	IC_{25}	Polley Lake	1	0	>100
			Quesnel Lake	1	0	
			Quesnel River	1	0	
	7-d survival and growth of rainbow trout	IC_{25}	Polley Lake	3	0	>100
			Quesnel Lake	2	0	
			Quesnel River	3	0	

n = number; v/v = volume per volume; h = hour; LC_{50} = median lethal concentration; NA = not applicable; d = day; IC_{25} = 25th percentile inhibitory concentration; QA/QC = quality assurance and quality control.

(a) includes discharge pumped from Polley Lake.

LOE	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Early life stage fish toxicity testing	Protect fish from reductions in survival, growth and development as a result of direct contact with water with elevated metals as a result of tailings deposited to the management area.	Conduct 96-h acute rainbow trout survival tests on field collected water samples.	Median survival is not reduced by more than 50% relative to control samples.	Polley Lake	Met decision criterion
				Quesnel Lake	
				Quesnel River	
		Conduct 7-d rainbow trout survival and growth toxicity tests on field collected water samples.	Performance is not inhibited by more than 25% relative to control samples.	Polley Lake	Met decision criterion
				Quesnel Lake	
				Quesnel River	
		Conduct 31-d rainbow trout survival and development toxicity tests on field collected water samples.		Quesnel River	Met decision criterion
		Conduct 7-d fathead minnow survival and growth toxicity tests on field collected water samples.		Polley Lake	Met decision criterion[a]
				Quesnel Lake	
				Quesnel River	

LOE = line of evidence; h = hour; % = percent; d = day.

(a) Anomalous toxicity observed in minority of samples in Polley Lake and Quesnel River in summer 2014 excluded from decision based on lack of validation in subsequent testing.

4.5.2 Tissue Concentrations in Prey Items

The assessment of bioaccumulation in aquatic organisms provides a line of evidence for multiple aspects of the risk assessment, including direct effects to the organisms themselves, and trophic transfer risk to species that forage on these food items. This section focuses on the potential for incremental risk to fish, considering the identification of differences between metals bioaccumulation in prey items at exposed versus reference areas (i.e., as a measure of relative exposure), but also considering the tissue burdens in relation to concentration-based benchmarks for dietary exposure to fish. Potential for incremental risk to benthic invertebrates and zooplankton through bioaccumulation pathways is discussed in Section 4.7.3 and Section 4.4.2.3, respectively.

In order to evaluate potential reductions in survival, growth, or reproduction of fish as a result of direct contact or ingestion of prey items (Assessment Endpoint 4; Section 2.4.7), measured concentrations of metals in field-collected samples of prey items such as benthic invertebrates, zooplankton, and small-bodied fish from exposure areas were compared to concentrations from reference areas to determine if there was any incremental exposure. Tissue concentrations of metals in prey items were separately compared to concentrations in diet associated with adverse effects in fish (i.e., dietary thresholds).

4.5.2.1 Study Design and Methodology

Zooplankton samples for tissue chemistry analysis were collected between 2014 and 2016 from Quesnel Lake and between 2015 and 2016 from Polley Lake by MPMC staff following methods outlined in Appendix M-1 Sampling methods differed between Quesnel Lake and Polley Lake to be consistent with previous data collection methods and thereby allow comparison with previous years' data sets. Samples were collected from three stations in Quesnel Lake: one exposed station called Hazeltine (QUL-Zoo-1; in the West Basin) and two reference stations, Horsefly (QUL-Zoo-7; near the Horsefly River) and Junction (QUL-Zoo-8; in the Main Basin where the East and North Arms meet). Samples were also collected from two stations in Polley Lake: P1 and P2. A more detailed description of the methods and sampling locations is provided in Appendix M-1. Data from the zooplankton monitoring sampling events are available in Appendix M-1 (2015 data), and Appendix M-2 (2016 data). Zooplankton tissue samples from the sampling events were analyzed for a suite of metals; however, only copper was the only constituent identified as a COPC in the problem formulation (Section 2.4.2). Other constituents have previously been evaluated that have potential to exhibit strong bioaccumulation (e.g., mercury, arsenic, selenium); however, as outlined in Appendix M-1 none of these substances exhibit consistent spatial trends in zooplankton tissue accumulation in Quesnel Lake or Polley Lake.

Zooplankton tissue concentrations of copper were plotted by sampling period for each station in Quesnel and Polley Lakes. The magnitude of potential effects due to the TSF embankment breach was determined by comparing copper concentrations in zooplankton tissue collected from exposed areas and reference areas. The concentrations of copper in zooplankton collected from reference areas of Quesnel Lake are considered representative of the natural range of variability, referred to as the normal range. Data from the two reference stations in Quesnel Lake (Horsefly and Junction) and all years of post-breach data (i.e., 2014 to 2016) were used to develop the normal range. Reference data were not available for Polley Lake and therefore the normal range of the Quesnel Lake reference data were used for comparison to Polley Lake exposed data.

A non-parametric method was used to calculate the normal range which used the 2.5th and 97.5th percentiles as the lower and upper limits (Barrett et al. 2015). Before calculating the normal range, the reference area data set was screened for inaccurate entries and potential outliers. To identify outliers in the reference data a common non-parametric method using the lower and upper fences of the data was adopted (Tukey 1977)[45]. Three outliers were identified in the pooled reference data (two samples collected at Horsefly (July and August 2016) and one sample collected at Junction (July 2016) using this method and removed from the data set used to calculate the normal range.

The median concentration of copper from the exposed area of Quesnel Lake (i.e., Hazeltine Channel mouth) and Polley Lake (P1 and P2 combined) was compared to the range observed in the reference location. Similar to the calculation of the normal range above, the data set was first screened for outliers. One outlier was identified in each of the two exposed data sets, and these anomalies were removed from the data sets used to calculate the median concentration.

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[45] The lower fence is defined as the first quartile minus 1.5 times the interquartile range (IQR) and the upper fence is defined as the third quartile plus 1.5 times the IQR. Observations that are beyond the upper and lower fences of the pooled reference data (i.e., all years combined) used to define the normal range were identified as outliers. The removal of identified outliers reduces the variability within the reference area data, thus narrowing the normal range and resulting in a more accurate and conservative estimate of the background variation.

In addition to comparing zooplankton tissue concentrations of copper in exposed areas to reference areas, tissue concentrations in the exposed areas were also compared to concentrations in diet associated with adverse effects in fish. There is no known dietary copper guideline or criterion for tissue consumption by fish, and therefore the dietary threshold for copper was derived from a review of primary literature. The literature search was limited to a web search using keywords (e.g., “dietary”, “copper”, “aquatic”, “toxicity”) and a database query of US EPA (2017) ECOTOX. Clearwater et al. (2002) compiled exposure concentrations of dietary copper and effects to fish that were reported in the literature. Of the fish species tested and compiled in Clearwater et al. (2002), rainbow trout was the only relevant species to Quesnel and Polley Lakes. Dietary exposure copper concentrations for rainbow trout in these documented toxicity tests ranged from 150 to 10,000 mg/kg dw, and the threshold for dietary copper toxicity to rainbow trout was approximately 664 to 730 mg/kg dw (Clearwater et al. 2002). The highest concentration below the threshold range that did not cause an effect to growth or survival (i.e., “no effect concentration”) was 500 mg/kg dw, which was tested in two independent studies using juvenile rainbow trout in long-term exposures (Knox et al. 1984; Handy et al. 1999). Other literature studies to those cited in Clearwater et al. (2002) did not report adverse effects to relevant freshwater fish at tissue concentrations lower than 500 mg/kg dw (Kamunde et al. 2002; Kamunde and Wood 2003; Kjoss et al. 2005). Therefore the no effect concentration of 500 mg/kg dw was used as a dietary threshold for copper in tissue.

4.5.2.2 Summary of Findings

The normal range (i.e., within the 2.5th to 97.5th percentiles) of the two reference stations in Quesnel Lake (Horsefly and Junction) from 2014 to 2016 was 14 to 49 mg/kg dw. The normal range is represented as the grey bar in the time-series plots of copper concentrations in zooplankton tissue samples collected from Quesnel Lake (Figure 34) and Polley Lake (Figure 35).

Quesnel Lake

No consistent temporal trends were observed in copper concentrations of zooplankton tissue measured from 2014 to 2016 in Quesnel Lake. In 2014 and 2015, copper concentrations in zooplankton were generally higher at Hazeltine compared to the other two stations (Figure 34). However, the pattern observed in 2016 was different, as concentrations in tissues from reference areas of the lake were elevated relative to the same stations in in 2014 and 2015, and also higher than observed at Hazeltine in 2016. This suggests that other factors not related to the TSF breach can influence copper uptake in zooplankton. Several individual copper measurements from tissue collected at Hazeltine were above the upper limit of the normal range, although the majority of these were within approximately 20% of the upper limit and therefore considered marginal (i.e., within the limit of analytical precision for copper in tissue samples [40%]). Two samples collected from Hazeltine exhibited tissue concentrations greater than 20% of the upper limit of the normal range. One elevated copper concentration of 254 mg/kg dw was observed at Hazeltine in May 2015, but it appeared to be anomalous (Appendix M-1) and was identified as an outlier in the distribution of the pooled exposed data (i.e., all years combined at Hazeltine). The other elevated copper concentration of 77 mg/kg dw was from the laboratory duplicate of the July 2016 sample and was approximately 60% greater than the upper limit of the normal range. The parent sample concentration was 45 mg/kg dw, which was within the normal range. In 2016, concentrations at both Hazeltine and the reference stations (Horsefly and Junction) appeared to be elevated compared to 2014 and 2015; however, the RPD for copper between the parent and duplicate sample in the 2016 laboratory certificate of analysis exceeded the laboratory data quality objective of 40% (Appendix M-3). This laboratory RPD failure indicates elevated uncertainty in the precision of copper concentrations in the whole batch, which included all 2016 zooplankton tissue collected from Quesnel and Polley Lakes. The three outliers identified in the pooled reference data and removed from the data set used to calculate the normal range were from the four samples collected at Horsefly and Junction in 2016.

The median concentration of copper in tissue samples collected from Hazeltine between 2014 and 2016 (excluding the identified outlier of 254 mg/kg dw) was 44 mg/kg dw. With the outlier included the median concentration was 45 mg/kg dw. Overall, the distributions of copper concentrations in Quesnel Lake zooplankton indicate that there is an increase in tissue accumulation at Hazeltine relative to the far-field sampling locations, but also that this difference is small in magnitude relative to other sources of variation in the data distributions, as evident in the 2016 data set that yielded large increases relative to previous years.

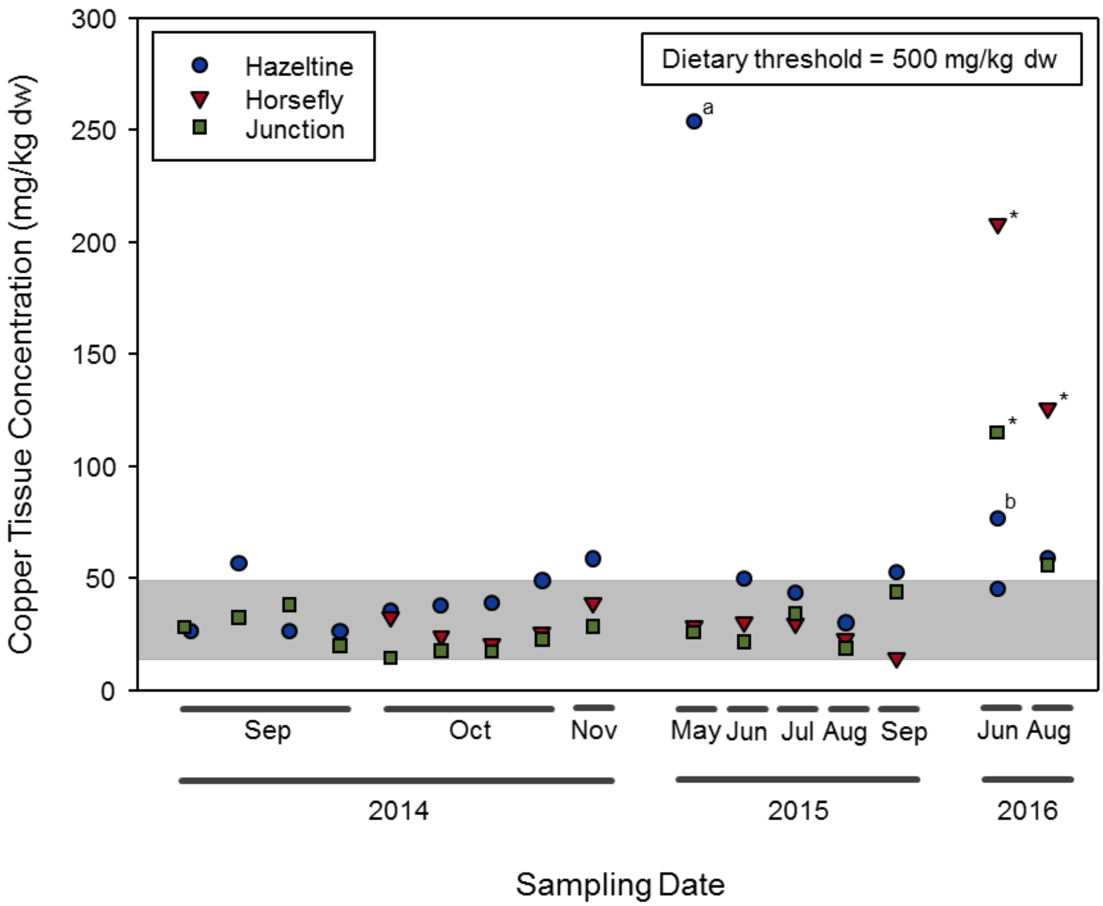

Figure 34: Concentrations of Copper in Zooplankton Tissue in Quesnel Lake

Note: The grey band is the normal range (i.e., 2.5th to 97.5th percentile) copper concentrations in zooplankton tissue collected from reference areas of Quesnel Lake (Horsefly and Junction) using all years (2014-2016) of post-breach data excluding statistical outliers (see reference data points with asterisks).

(a) Elevated copper concentration of 254 mg/kg dw observed at Hazeltine in May 2015 appeared to be anomalous (Appendix M-1) and was identified as an outlier in the distribution of the pooled exposed data.

(b) The relative percent difference for copper between duplicate samples in the 2016 laboratory certificate of analysis failed the laboratory data quality objective indicating uncertainty in the precision of copper concentrations in the whole batch. Due to the uncertainty in the precision of copper concentrations, both the parent and duplicate sample concentration (July 2016 data point with asterisk) were provided in the figure.

Polley Lake

Zooplankton tissue concentrations of copper were generally similar between sampling stations (P1 and P2) in Polley Lake during the 2015 sampling events and in July 2016, with the exception of the September 2015 sampling event when P1 was higher than P2 (Figure 35). Copper concentrations exhibited a general increasing trend from May to September 2015, which appeared to continue in July 2016. In August 2016, the concentration trends for copper in P1 and P2 diverged, with the concentration at P2 (128 mg/kg dw) two orders of magnitude higher than the concentration at P1 (4.6 mg/kg dw). As discussed above, the RPD for copper between duplicate samples in the 2016 laboratory certificate of analysis failed the laboratory data quality objective indicating elevated uncertainty in the precision of copper concentrations in the whole batch. Therefore the results of the 2016 analysis are questionable.

The median concentration of copper in tissue samples collected from P1 and P2 in 2015 and 2016 (excluding the identified outlier) was 35 mg/kg dw. With the outlier included the median concentration was 37 mg/kg dw.

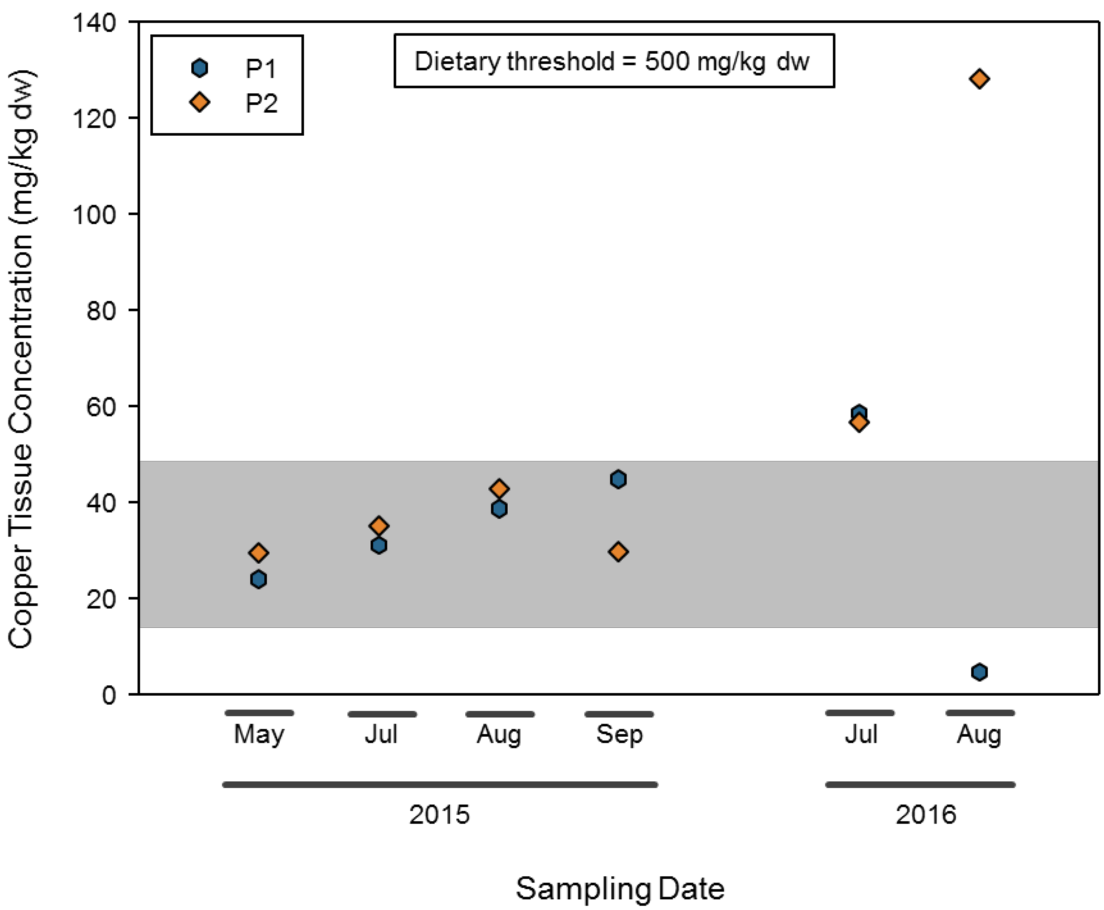

Figure 35: Concentrations of Copper in Zooplankton Tissue in Polley Lake

Note: The grey band is the range of 2.7th to 97.5th percentile copper concentrations in zooplankton tissue collected from reference areas of Quesnel Lake (Horsefly and Junction) using all years (2014-2016) of post-breach data. This information is provided for comparison purposes only.

4.5.2.3 Confounding Factors

This particular line of evidence is focused on chemical stressors only. The influence of the physical burial processes on the zooplankton bioaccumulation processes are not expected to be substantial given that the exposure occurs in the water column rather than the sediment bed. The TSF embankment breach and subsequent changes to water quality in Quesnel and Polley Lakes did not result in large differences in copper accumulations in zooplankton tissue between exposed areas and reference areas, and therefore the relative contributions of physical and chemical stressors did not require assessment. It is possible that variations in total suspended solids in the lake (and the source of these solids vis-à-vis the distribution of settled tailings) could influence the bioaccumulation of copper into zooplankton, by affecting the concentration and bioavailability of copper associated with the particulate fraction of suspended solids. However, the data collected to date indicate that the variation in these processes is small in relation to other sources of variance in the bioaccumulation data over space and time.

4.5.2.4 Uncertainty Analysis

Measuring metal concentrations in prey items provides information relevant to understanding whether metals are bioaccumulating or biomagnifying in the food chain. Comparison of metal concentrations among the management and reference areas evaluates the relative increased exposure of zooplankton whereas comparison to dietary thresholds evaluates potential effects to fish through dietary exposure. Potential sources of uncertainty in the line of evidence are:

- Methodologies for collection and analysis of tissue chemistry are well-established, with accepted QA/QC measures and standardized data analysis methods.
- The RPD for copper between duplicate samples in the 2016 laboratory certificate of analysis exceeded the laboratory data quality objective of 40%. ALS attributed the laboratory data quality objective failure to sample heterogeneity. The laboratory homogenizes the whole sample and then two aliquots are taken from the whole sample for analysis. During laboratory analysis, homogenization of the sample may not have been complete (e.g., debris in the gut contents that was not broken up), such that subsampling for analysis resulted in variability in the results. This laboratory RPD failure indicates elevated uncertainty in the precision of copper concentrations in the whole batch, which included all 2016 zooplankton tissue collected from Quesnel and Polley Lakes.
- There are no officially promulgated "safe" levels for dietary copper concentrations for fish. The dietary benchmark was based on dietary effects concentrations found in available literature, and was derived using a no-effect concentration. Tissue concentrations causing effects to fish were reported at higher concentrations, so the dietary threshold is likely a conservative concentration.
- Comparison of tissue chemistry of zooplankton in the management area to reference areas (i.e., background conditions) provides context for potential breach-related impacts. Changes in tissue chemistry of prey items relative to background is a measurement of exposure and does not necessarily mean that an adverse effect is occurring at the individual or population level of consumers (i.e., fish). However, the second assessment endpoint (i.e., comparison to dietary threshold) allows comparison of tissue chemistry concentrations to concentrations associated with adverse effects. The tissue-based benchmark provides a high degree of certainty that concentrations less than the guideline are unlikely to present a risk to fish through dietary ingestion.

- Some fish species have wide-ranging territories and may not exclusively be feeding on prey items from a particular area. It is assumed that the characteristics at Hazeltine provide a reasonable worst-case exposure condition given the proximity to the source of the tailings release and the likelihood that most fish would aggregate their exposures over broader areas including those with lower tailings influence.

4.5.2.5 Comparison to Decision Criteria

For the prey tissue chemistry line of evidence, the original decision criteria proposed by Golder to determine effects as a result of the breach were:

- average concentration in zooplankton samples collected from the management area is not more than 20% higher than the average concentration in samples collected from reference areas
- concentrations in prey items are less than dietary thresholds for adverse effects

These criteria were re-evaluated based on the temporal and natural variability in concentrations observed at reference areas and were revised to:

- median concentration in zooplankton samples collected from the management area is lower than the upper limit of the normal range of samples collected from reference areas
- median concentrations of prey items are less than the dietary threshold for adverse effects

Quesnel Lake

The median copper concentration of zooplankton tissue samples collected from Hazeltine sampling station between 2014 and 2016 (excluding the identified outlier of 254 mg/kg dw) was 44 mg/kg dw, which is below the upper limit of the normal range (49 mg/kg dw). With the outlier included the median concentration was 45 mg/kg dw, which is still below the upper limit of the normal range. The concentrations of copper in zooplankton tissue from the exposed area of Quesnel Lake do not appear to be different from the concentrations observed in the reference area, resulting in a “no difference” for this measurement endpoint.

Copper concentrations in zooplankton tissue collected from the exposed area of Quesnel Lake ranged from 26 to 254 mg/kg dw, which were below the dietary threshold of 500 mg/kg dw. This resulted in a “no effect” for this measurement endpoint.

Overall, there were no discernable differences in copper concentrations of zooplankton tissue samples between the exposed area and the reference areas of Quesnel Lake, which resulted in “no change” for this line of evidence.

Polley Lake

The median concentration of copper in tissue samples collected from P1 and P2 in 2015 and 2016 (excluding the identified outlier) was 35 mg/kg dw, which is below the upper limit of the normal range (49 mg/kg dw). With the outlier included the median concentration was 37 mg/kg dw, which is still below the upper limit of the normal range. The concentrations of copper in zooplankton tissue from Polley Lake do not appear to be different to the concentrations observed in the reference area of Quesnel Lake, resulting in a "no difference" for this measurement endpoint.

Copper concentrations in zooplankton tissue collected from Polley Lake ranged from 4.6 to 128 mg/kg dw, which were below the dietary threshold of 500 mg/kg dw. This resulted in a "no effect" for this measurement endpoint.

Overall, there were no discernable differences in copper concentrations of zooplankton tissue samples between Polley Lake and the reference areas of Quesnel Lake, which resulted in "no change" for this line of evidence.

LOE	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Zooplankton tissue chemistry	Protect fish from reductions in survival, growth or reproduction as a result of ingesting zooplankton with elevated concentrations of metals in their tissues.	Measure the concentration of copper in field-collected samples of zooplankton.	The median concentration in samples collected from the management area is lower than the upper limit of the normal range of samples collected from reference areas.	Quesnel Lake	No difference
				Polley Lake	No difference
		Compare concentrations of metals in field-collected samples of zooplankton to concentrations in diet associated with adverse effects in fish.	Median concentration in zooplankton tissue from the management area is less than dietary threshold for adverse effects.	Quesnel Lake	No effect
				Polley Lake	No effect
Benthic invertebrate tissue chemistry	Protect fish from reductions in survival, growth or reproduction as a result of ingesting benthic invertebrates with elevated concentrations of metals in their tissues.	Measure the concentration of metals in field-collected samples of benthic invertebrates.	See Section 4.7.3		
		Compare concentrations of metals in field-collected samples of benthic invertebrates to concentrations in diet associated with adverse effects in fish.	Median concentration in benthic invertebrate tissue is less than dietary threshold for adverse effects.	Quesnel Lake	No effect
				Polley Lake	No effect

LOE	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Small-bodied fish tissue chemistry	Protect fish from reductions in survival, growth or reproduction as a result of ingesting small-bodied fish with elevated concentrations of metals in their tissues.	Measure the concentration of metals in field-collected samples of small-bodied fish.	See Section 4.5.3		
		Compare of concentrations of metals in field-collected samples of small-bodied fish to concentrations in diet associated with adverse effects in fish.	Median concentration in small-bodied fish is less than dietary threshold for adverse effects.	Quesnel Lake	No effect
				Polley Lake	No effect

4.5.3 Tissue Concentrations in Fish

Assessment of differences in metal tissue concentrations of fish among exposed and reference areas helps to clarify whether exposure to and subsequent uptake of metals in fish have increased as a result of the TSF embankment breach. If elevated fish tissue concentrations are occurring as a result of the breach, concentrations can be compared to effect-based tissue benchmarks to evaluate the potential for toxicity to fish.

4.5.3.1 Study Design and Methodology

Fish tissue chemistry data collected subsequent to the TSF embankment breach are reported in Appendix D. Multiple stakeholders collected fish tissue samples in 2014 and 2015, and sampling was generally opportunistic. The data were divided into fish collected from exposed and reference areas, and subdivided by species, tissue type, and year to facilitate comparisons. Basic descriptive statistics were calculated and boxplots were generated to facilitate comparisons among exposed and reference areas. Data were compared to available tissue guidelines (arsenic, mercury, and selenium) primarily for the purpose of screening for human health.

Data were also collected by Minnow in 2015 and 2016. These data are preferred for the ecological risk assessment because the 2015-2016 data were collected based on *a priori* sampling plans that are more balanced in terms of species collected, areas sampled, and number of samples. Sampling locations and sample collection methodology are summarized below, and details are provided in Appendix D-2. In brief:

- Fish were collected from Polley Lake and Quesnel Lake, plus several reference lakes and/or their tributaries, using a variety of collection methods (e.g., gillnets, tangle nets, trap nets, hoop nets, angling).
- Target fish species (sentinel species) were selected based on their presence at the potentially impacted areas, and considering their recreational value (sportfish: rainbow trout *Oncorhychus mykiss*, lake trout *Salvelinus namaycush*, kokanee *Oncorhynchus nerka*, burbot *Lota lota*) or their benthic feeding habitat and small home ranges (peamouth chub *Mylocheilus caurinus*, northern pikeminnow *Ptychocheilus oregonensis*, redside shiner *Richardsonius balteatus*, largescale sucker *Catostomus macrocheilus*, longnose sucker *C. catostomus*). Only fish of appropriate size were retained for processing, all other fish were released.

- For each sampling area, a minimum sample size of eight fish for each species was targeted to facilitate statistical comparisons. Retained fish were measured for length and weight and aging structures (otoliths, scales, or pectoral fin rays) were collected from each fish. Peamouth chub, northern pikeminnow, and redside shiner were typically submitted as whole-body samples. For other fish species samples of internal organs (liver, kidney, and ovary) were collected along with a boneless, skinless muscle sample.
- Tissue samples were analyzed by ALS Environmental for metals by collision cell inductively coupled plasma-mass spectrometry (CRC-ICP-MS) or high resolution inductively coupled plasma spectrometry (HR-ICP-MS), mercury by cold vapour atomic fluorescence spectrophotometry (CVAFS), and moisture content following standard analytical protocols.
- Summary statistics were reported for the reference and exposed areas for each species/tissue type/area comparison. Data were plotted for comparisons among exposed and reference areas and included some pre-breach data from Polley Lake collected in 2010. No species-specific reference data were available for comparison to kokanee and rainbow trout collected in Quesnel Lake and Quesnel River in 2015, which precluded spatial comparison of exposed and reference conditions.

4.5.3.2 Comparison to Decision Criteria

The decision criterion for the fish tissue chemistry line of evidence considered the following:

- First, whether there was a substantive increase[46] in tissue concentrations of metals in fish from exposed areas relative to reference areas.
- Second, whether an observed increase is biologically relevant based on comparison to tissue residue guidelines, if available, or TRV.

4.5.3.3 Summary of Findings

Detailed results of tissue metal analysis are provided in Appendix D-2. Only copper and arsenic are discussed below as these were identified as COPCs in water (copper) and sediment (arsenic and copper).

Polley Lake

Fish tissue chemistry data collected in Polley Lake in 2015 (tributary Frypan Creek) and 2016 were compared to Polley Lake data collected in 2010 (pre-breach) and to data collected from two reference lakes in 2015 (Bootjack Lake and Trio Lake). Reference lakes have naturally different geochemistry than Polley Lake, which may influence fish tissue chemistry and confound comparisons between lakes. For example, tissue concentrations of arsenic for Trio Lake were up to 1.7-times higher than those from Bootjack Lake and fish from both of these reference lakes had lower tissue arsenic concentrations relative to pre-breach conditions in Polley Lake. Fish from Trio Lake had lower muscle concentrations of copper than pre-breach, whereas fish from Bootjack Lake had concentrations 1.3-times higher than pre-breach, again likely reflecting differences in geochemistry between lakes.

[46] The specification of whether a substantive increase occurred depends on the results of comparisons of fish tissue concentrations from exposed areas to both reference lakes and pre-breach conditions, where available, along with consideration of the observed magnitudes of difference observed, the sample sizes available for comparison, and the consistency in relationships among tissue types. These contrasts were not amenable to simple quantitative decision rules or statistical significance tests.

The results for arsenic accumulation in Polley Lake fish were:

- Rainbow trout—Mean concentrations in muscle in 2015 and 2016 were up to 4.8-times higher than those collected from the two reference lakes in 2015. However, the 2015 muscle concentrations were lower than pre-breach data and 2016 concentrations were similar to (1.3-times higher) pre-breach (Figure 5 of Appendix D-2); the divergence of these findings highlights the uncertainty associated with making specific statistical comparisons. Measured liver concentrations in Polley Lake were up to 6.6-times higher and kidney concentrations were similar (1.2-times higher) relative to reference; no pre-breach data were available for these tissue types. Ovary concentrations were similar to reference (i.e., up to 1.3-times higher), but were lower than pre-breach conditions.
- Longnose sucker—Muscle concentrations in longnose sucker collected in 2016 were 1.8-times higher than pre-breach; no reference lake data were available for this species; ovary concentrations were similar to pre-breach.

The results for copper in Polley Lake fish were as follows:

- Rainbow trout—Muscle concentrations were up to 3.1-times higher than in reference lakes and 1.6-times higher than pre-breach (Figure 6 of Appendix D-2); liver concentrations were up to 3.3-times higher than the Bootjack Lake reference, but were lower than the Trio Lake reference and kidney concentrations were similarly lower than the Trio Lake reference. No pre-breach liver or kidney data were available; ovary concentrations were lower than reference, and similar to (1.3-times higher) pre-breach.
- Longnose sucker—muscle tissue was 1.6-times higher than pre-breach; ovary concentrations were similar to pre-breach.

Copper is an essential metal and well-regulated in fish (Grosell 2012); modest increases in certain tissue types in a given season do not necessarily reflect increased chronic exposure and uptake, particularly when the pattern is not consistent among tissue types or species. Overall, the tissue results for fish in Polley Lake suggest modest (i.e., less than two-fold) increases in arsenic and copper concentrations relative to pre-breach concentrations.

Quesnel Lake

Fish tissue chemistry data collected in the West Basin of Quesnel Lake near the mouth of the Hazeltine Channel in 2015 and 2016 were compared to data collected for the same species from reference areas of the lake (e.g., East Arm, near Grain Creek) in the same years. This resulted in between area comparisons for six different species of fish: burbot, lake trout, largescale sucker, peamouth chub, redside shiner, and northern pikeminnow.

Fish collected near the mouth of the Hazeltine Channel had mean concentrations of arsenic up to 2.7-times higher in muscle or whole-body and up to 1.9-times higher in liver relative to same species collected from reference areas in Quesnel Lake (Figure 12 of Appendix D-2). Lake trout from exposed areas had mean arsenic concentrations 1.8-times higher in both kidney and ovary compared to the reference area.

Similarly, mean concentrations of copper were 1.5-times higher in muscle or whole-body and 1.9-times higher in liver, in fish from exposed versus reference areas (Figure E.4 of Appendix D-2). Lake trout from exposed areas had mean copper concentrations 1.5-times higher in kidney and 1.7-times higher ovary compared to the reference area; however, there was one very high kidney concentration in the exposed fish that strongly influenced this comparison and sample size was limited for ovaries.

Overall, these results suggest modest increases in arsenic and copper concentrations in tissue of fish from the West Basin relative to reference areas. Most of the pairwise comparisons between exposed and reference areas indicated changes of less than a factor of two, similar to the findings reported above for Polley Lake.

Comparison to Tissue-Residue Guidelines or Thresholds

There are no tissue residue guidelines from BC MoE for copper or arsenic. A literature search for TRVs was not undertaken; instead Golder relied upon an existing compendium of tissue residues associated with biological effects in aquatic organisms, as prepared by Jarvinen and Ankley (1999). Toxicity reference values, expressed as a tissue residue concentration, were identified for coldwater fish species based on ecological relevance to the Mount Polley mine area. TRVs were not selected if a study considered mixed exposure to water and sediment or the organic form of metal (e.g., for arsenic).

Metal uptake varies among fish species and tissue types, and is dependent upon source (i.e., exposure from the water, food, or both), environmental conditions (e.g., water temperature, pH, hardness), exposure frequency, exposure duration (i.e., acute or chronic), and tissue function (e.g., storage versus elimination capacity). Some metals are known to accumulate to a greater degree in specific tissues and in some instances, organ-specific samples are more relevant for evaluating exposure and potential for direct effects to fish than muscle tissue or whole-body samples. Tissue concentrations of arsenic and copper were compared to tissue-specific TRVs where available, as summarized below:

- **Arsenic**—A whole-body concentration of 10–17 mg/kg dw (2.0–3.4 mg/kg ww) was associated with a no-observable effects on survival and growth of juvenile rainbow trout in a 77-day aqueous exposure to 18 mg/L (as sodium arsenate; McGeachy and Dixon 1990). Arsenic concentrations in muscle or whole-body of fish collected from all areas were below this TRV. No other organ-specific TRVs for arsenic were reported for coldwater fish. Aqueous concentrations of arsenic in both Polley and Quesnel lakes did not exceed the ambient WQG following the breach, providing confirmation of the lack of significant risk through water column exposure.

- **Copper**—A muscle concentration of 2.5 mg/kg dw (0.5 mg/kg ww) was associated with no effects on survival of adult rainbow trout following an 8-hour aqueous exposure to 0.1 mg/L (as copper sulphate) and 1 week of post-exposure observation (Handy 1992). This was the only TRV reported by Jarvinen and Ankley (1999) for muscle or whole-body. Muscle or whole-body concentrations in both small-bodied fish and larger sportfish from exposed and reference areas exceeded (but were less than 2-times higher than) this tissue residue value. However, this study represents an acute exposure to a copper concentration up to 50-times higher than the ambient WQG and it is not representative of site-specific environmental conditions, as aqueous concentrations of copper in Polley and Quesnel lakes were consistently below BC WQGs through 2015 and 2016. This highlights the challenges associated with developing whole body TRVs for essential substances that are internally regulated by organisms.

- Liver and kidney are considered the primary sites of accumulation or toxicity of copper (Appendix D-1). In the absence of a relevant muscle/whole-body TRV, organ-specific TRVs were also reviewed and considered more appropriate.
  - Copper concentrations of 180 mg/kg dw (36 mg/kg ww) in liver and 7.5 mg/kg dw (1.5 mg/kg ww) in kidney were associated with reduced growth in juvenile rainbow trout in a 105-day aqueous exposure to 0.07 mg/L (as copper acetate; Buckley et al. 1982). This duration of study is more representative of chronic exposure that would occur under environmental conditions; however, the exposure concentration was 35-times the chronic WQG. In another study, concentrations of 240 mg/kg dw (48 mg/kg ww) in liver and 17 mg/kg dw (3.3 mg/kg ww) in kidney were associated with no effects on survival, growth, and reproduction in brook trout (*Salvelinus fontinalis*) in a 720-day aqueous exposure to 0.0094 mg/L (as copper sulphate: McKim and Benoit 1971; 1974). This exposure concentration is still almost 5-times the chronic WQG and 2–10-times the average aqueous concentrations measured in Polley and Quesnel lakes through 2015 and 2016. Liver and kidney concentrations in fish collected from exposed areas did not exceed these organ-specific TRVs, apart from kidney concentrations in lake trout (7.8 mg/kg dw) that approximated the lower TRV.
  - Copper concentrations of 600 mg/kg dw (102 mg/kg ww) in liver and 23 mg/kg dw (4.5 mg/kg ww) in kidney were associated with no effects on growth in juvenile rainbow trout in a 56-day dietary exposure to 287 mg/kg in food (as copper sulphate; Lanno et al. 1985). This dietary exposure level was the lowest dietary concentration of that study and is 1.6-times higher than copper concentrations in benthic invertebrates from the profundal near-field area of Quesnel Lake (mean = 178 mg/kg dw). Tissue concentrations in fish from all areas were below these TRVs.

4.5.3.4 Uncertainty Analysis

The internal regulation of essential elements such as copper in aquatic organisms introduces uncertainty to the comparisons of tissue budens among locations (and over time) as an indicator of exposure and risk to fish. Other lines of evidence (e.g., dietary tissue chemistry) provide indirect estimates about how metals are likely to accumulate in the food chain from sediment; however, measurement of tissue concentrations provides a more direct approach to assessing potential risks to fish. Metal concentrations can be compared between the management and reference areas to evaluate the relative increased in exposure of fish. Tissue concentrations can also be compared to BC MoE tissue residue guidelines or thresholds from the literature, although the availability and technical basis of these thresholds can be limited.

Fish were collected and processed following well-established field methods, using consistent approaches across sampling events. Metal concentrations in tissues were measured using standardized analytical test methods with robust QA/QC practices. Minnow completed a data quality assessment upon receipt of chemistry results and determined that results met data quality objectives and was considered to be acceptable for the use of interpretation and derivation of conclusions.

If sufficient numbers of samples are collected, there is often a high degree in reliability in the statistical analysis that can be used to evaluate representativeness. However, in some cases sample size was limited, particularly in historic and reference data sets. The same species were not necessarily present or caught in reference areas (e.g., longnose sucker not in Bootjack and Trio lakes), which precluded spatial comparisons of exposed and reference areas. Comparison to historical data for Polley Lake provided context for potential breach-related impacts, as there are differences in geochemistry between lakes that introduce some natural variability in comparisons to reference lakes.

Tissue-based guidelines provides a high degree of certainty that concentrations less than the guideline are unlikely to present a risk. TRVs that represent no-effects concentrations have a high degree of certainty that concentrations less than the TRV are unlikely to present a risk.

Differences in tissue chemistry among locations or relative to reference does not necessarily mean that an adverse effect is occurring at the individual or population level. For such measures of exposure, and particularly given the internal regulation of copper by fish, the strength of association to the assessment endpoint is considered low. Potential for toxicity to fish from aqueous exposure was evaluated directly in chronic toxicity tests discussed in Section 4.5.1.

Overall, the fish tissue results indicate modest increases in arsenic and copper tissue concentrations in Polley Lake (within a factor of two) relative to pre-breach concentrations and similar modest increases in fish from the West Basin of Quesnel Lake relative to reference areas. The number and type of tissue samples, in addition to constraints associated with confounding factors (e.g., differences in fish size, geochemical differences among water bodies) do not allow for rigorous statistical comparisons or calculation of precise rates of change.

In spite of the small increases in tissue concentrations, these observed increases are not considered to be biologically relevant, as tissue concentrations were typically below tissue/organ-specific TRVs in which exposure concentrations are higher than those present in Polley and Quesnel lakes.

4.5.4 Fish Histology

4.5.4.1 Study Design and Methodology

Exposure to stressors of both natural and anthropogenic origin can adversely affect the immune system of fish, and therefore the ability to resist disease (Lloyd 1987). An increased prevalence of disease in a population could be indicative of prolonged exposure to stressors in the environment (Bly et al. 1997). Potential effects from the TSF embankment breach and subsequent changes to water quality on the incidence of disease in fish was assessed based on histological analyses of tissues from sockeye salmon (*Oncorhynchus nerka*) collected from Quesnel Lake (Appendix P-2). Juvenile sockeye were collected from the North Arm (*n*=20), West Arm (*n*=20), East Arm (*n*=20), and Middle Arm (*n*=21) of Quesnel Lake in October 2014.

Tissues relating to gill, liver, stomach, brain, cranium, and the integument were evaluated by a ~~qualified~~ fish pathologist and abnormalities were reported (University of Prince Edward Island 2015; data courtesy of BC MoE; Appendix N-2). The presence of digesta in the stomach was also reported as evidence of active feeding, because diseased fish may reduce their food intake or cease to feed. The reported data were categorized based on the type of observation (i.e., active feeding or abnormality); abnormalities were then sub-categorized into damage received during sample collection, parasites, and previous trauma. The results of this analysis are included in Appendix P-1 and summarized in Table 35.

Results of the histological analyses were assessed qualitatively to identify notable differences, if any, in feeding and tissue abnormalities among the exposure (West Arm) and reference areas in Quesnel Lake.

4.5.4.2 Summary of Findings

Based on a qualitative assessment of the histological results for sockeye salmon collected from Quesnel Lake in 2014, digesta was present in nearly all fish samples which is indicative of an enduring population. Additionally, the incidence of disease and parasites was not found to be different in the West Arm when compared to other areas of Quesnel Lake, considered to be reference areas (Appendix P-2; Table 35). This suggests that the disease rates in fish were not linked to degree of tailings exposure. Therefore, the information on fish histology are not indicative of an ongoing water quality-induced stress associated with the tailings.

Table 35: Summary of Histological Observations in Sockeye Salmon Collected from Quesnel Lake, 2014

Histological Observation	Percent (%) of Fish Samples with Condition			
	West Arm (*n*=20)	**East Arm** (*n*=20)	**Mid Arm** (*n*=21)	**North Arm** (*n*=20)
Active feeding (digesta present in sample)	**95**	**95**	**90**	**100**
Trauma during sample collection and preservation (epidermal denuding)	**100**	**100**	**100**	**100**
Previous trauma during fish history (total)	**95**	**100**	**100**	**60**
Mono infiltration (liver trauma)	15	5	5	5
Microgranulomas (brain trauma)	40	30	24	10
Ulcerative epidermatopathy (cranial trauma)	100	100	100	60
Mononuclear lymphocytes (other trauma)	5	0	5	5
Parasites (total)	**80**	**70**	**48**	**90**
Sessile protozoans (gill parasite)	80	65	48	90
Encysted protozoan (*Icthophthirius sp*.; gill parasite)	5	0	0	0
Microsporidian xenoma (brain parasite)	5	5	10	5
Cestode in peritoneum (abdominal parasite)	0	0	0	5
Metazoan parasite induced (whole organism)	10	0	0	0
Xenoma in meninges (brain parasite)	0	5	0	0

4.5.4.3 Comparison to Decision Criteria

The fish histology information is intended to be considered in conjunction with other lines of evidence (e.g., fish metrics and habitat/productivity measures) and in the context of fish population and tissue chemistry assessments. Results of the histological analyses were qualitatively compared among exposure and reference locations in Quesnel Lake; the decision criterion was not met if the overall incidence of abnormalities was higher in the West Arm compared to reference areas (East Arm, Mid Arm, and North Arm).

Stressors of both natural and anthropogenic origin can adversely affect histological characteristics. An increased prevalent of disease in a fish population could be indicative of prolonged exposure to stressors in the environment; however, cause and effect relationships are uncertain. Evidence of histological changes can have multiple causes including sampling artifacts (e.g., fish handling) or natural stressors (e.g., background incidence of parasites). Therefore, increased abnormalities may not be an indication of a diseased state resulting from the TSF embankment breach. This line of evidence should be considered in combination with other lines of evidence, such as fish condition (Section 4.5.5). The strength of association to the assessment endpoint is considered moderate.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Fish histology	Protect fish from reductions in survival, growth and reproduction as a result of multiple exposure pathways.	Compare histology of juvenile sockeye salmon collected throughout Quesnel Lake in October 2014.	Overall frequency of abnormalities among sampling zones within Quesnel Lake are similar.	Quesnel Lake	Met decision criterion

4.5.5 Fish Condition Factors

Body metrics of fish (i.e. length, weight, and body condition) are commonly assessed as measures of growth and energy storage. Fulton's condition factor is a ratio between body length and body weight that is commonly used to assess the body condition of fish in fisheries assessments. Fish condition varies with a variety of factors, including habitat quality, food availability, seasonal temperatures, and contaminant loading in body tissues.

4.5.5.1 Study Design and Methodology

Information on fish collected from Polley Lake, Quesnel Lake, and Edney Creek was gathered from Appendices D-2, P-2, P-3 P-4, and P-5. In brief, body metrics data were available for rainbow trout in Polley Lake from 1973, 1995, and 2012 prior to the TSF embankment breach, and 2014 and 2016 following the breach. Similarly, data were available for longnose sucker in Polley Lake from 1995 and 2012 prior to the breach, and 2014 and 2016 following the breach. In Quesnel Lake, data were available for a variety of fish species collected in 2015 and 2016 in different regions of the lake. Sampling from Quesnel Lake generally included two groupings for a given species in a given year: fish collected from the West Basin of the lake in close proximity to the mouth of the Hazeltine Channel and fish collected from areas east of Cariboo Island (e.g., North Arm, East Arm), serving as a reference area. A summary of sampling methodologies, sample sizes, and fish body characteristics is provided in Appendix P-4. For Edney Creek, fish usage was characterized in the lower, reconstructed area of the creek relative to an unimpacted (reference) area upstream near the Horsefly-Likely Forest Service Road.[47] Results for Edney Creek are reported in Appendix P-5.

Fulton's condition factor (K) was calculated as:

$$K = 100\, x \frac{m}{l^3}$$

[47] See Figure 2 in Appendix P-5 for locations of fish collection in Edney Creek.

Where K = Fulton's condition factor; m = mass in grams; l = fork length in centimetres.

Condition factor was calculated for each fish collected and then temporal comparisons were made for fish collected in Polley Lake (i.e., pre- versus post-breach), spatial and temporal comparisons were made for fish collected in Quesnel Lake (i.e., West Basin versus reference area), and a spatial comparison was made for Edney Creek (reconstructed habitat versus upstream reference) via box plots and calculations of mean condition factor.

4.5.5.2 Summary of Findings

Polley Lake

Rainbow trout have been collected from Polley Lake in several sampling events spanning back to 1973. Mean condition factor was highest in 1973 and 1995, and lowest in 2012 prior to the breach. The mean condition factor in 2014 shortly after the breach was higher than that in 2012, and the condition factor in 2016 was similar to that in 2014 (Figure 36). Possible explanations for the decrease in condition factor compared to historical data are decrease over time in the amount of time spent fishing and the number of fish removed from Polley Lake, potentially leading to an increasing population density in the lake and increased competition for resources (Lirette 2015). Overall, the average body condition of rainbow trout in Polley Lake does not appear to have been influenced by the influx of material following the breach in 2014.

Longnose sucker have been collected from Polley Lake in monitoring programs spanning back to 1995. Mean condition factor was highest in 1995, followed by 2012. Mean condition factor in 2014 and 2016 following the breach were similar and within the range of historical data (Figure 37). Similar to rainbow trout, it is possible that condition factor of longnose suckers collected in 1995 was higher due to higher fishing activity and decreased population density. Overall, the average body condition of longnose sucker does not appear to have been influenced by the influx of material following the breach in 2014.

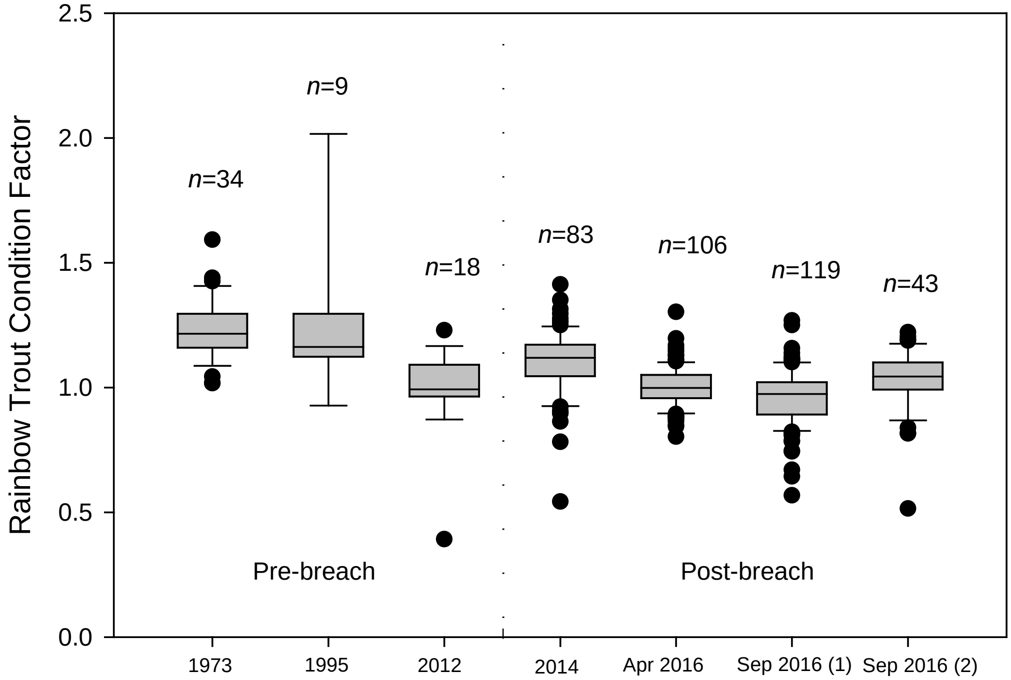

Figure 36: Condition Factors for Rainbow Trout Collected From Polley Lake, Pre-Breach (1973, 1995, 2012) and Post-Breach (2014, 2016)

Note: The first sampling event in September 2016 involved collection via hop nets and seine nets; the second sampling event in September 2016 involved collection via gillnets. Boxplots indicate minimum values, 25th, 50th, and 75th percentiles, maximum values, and outliers.

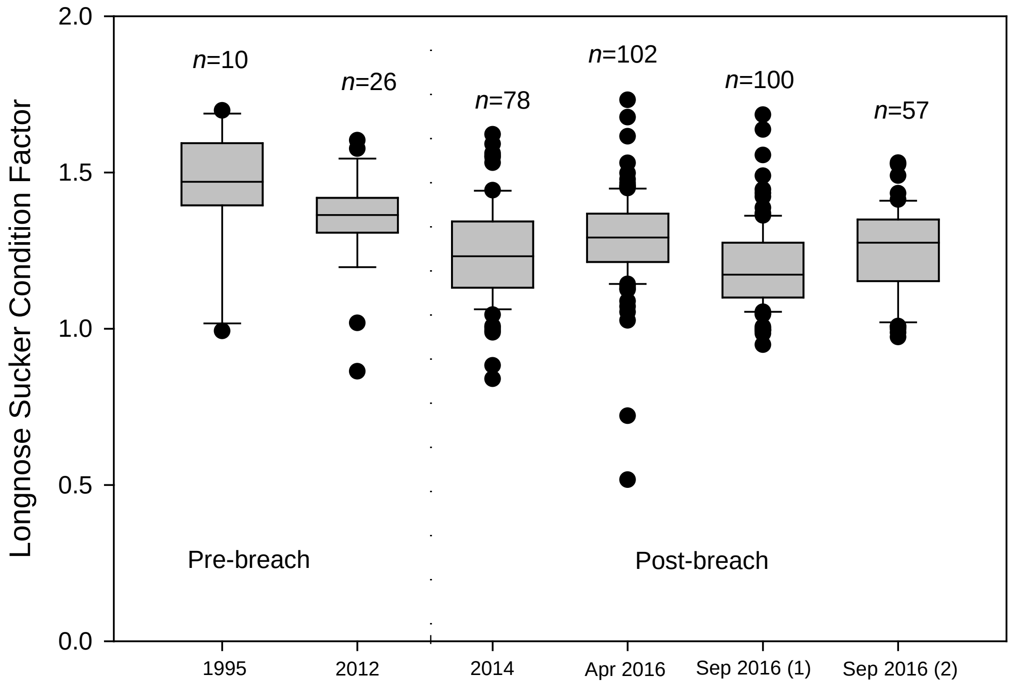

Figure 37: Condition Factors for Longnose Suckers Collected From Polley Lake, Pre-Breach (1995, 2012) and Post-Breach (2014, 2016)

Note: Refer to notes in Figure 36.

Quesnel Lake

Data were available for the following seven fish species that were collected during at least one sampling event between 2014 and 2016 following the breach: burbot, lake trout, largescale sucker, peamouth chub, redside shiner, northern pikeminnow, and sockeye salmon. With the exception of sockeye salmon, all other fish species were collected by Minnow and were targeted due to their importance as either sportfish or their small ranges and preference for benthic habitats and feeding environments (Appendix P-3). Juvenile sockeye salmon were collected by Fisheries and Oceans Canada (DFO) as part of an ongoing monitoring program and were included in this analysis due to the sensitivity of early life stages to contaminants, as well as the cultural, economic, and ecological importance of the species in the region.

In 2014, the mean condition factor of juvenile sockeye salmon was highest in those collected from the West Arm[48] of Quesnel Lake compared to other regions of the lake. This suggests that there was no adverse effect on foraging efficiency of juvenile sockeye salmon as a result of the breach. It is possible that the juvenile sockeye in the West Arm fed primarily near the surface where turbidity was lower compared to at depth and that foraging efficiency was not impacted due to abundance of zooplankton in this region. Alternatively, a region-specific decrease in foraging efficiency associated with the influx of tailings material may have been offset by increased food availability due to an influx of nutrients along with the tailings (Appendix M-2). Additional monitoring of juvenile sockeye in Quesnel Lake was performed by DFO in 2015, and this pattern did not persist. Condition factors in sockeye salmon in Quesnel Lake in 2015 were similar across all regions and also similar to those measured in 2014 (Figure 38).

Despite differences in sampling methodology, size distributions for largescale sucker were similar between years. Mean condition factors were similar both across years and regions for largescale sucker (Figure 39) and the data do not suggest a difference in body condition for largescale sucker residing in the West Basin compared to those in the reference areas (i.e., North Arm and East Arm).

For the other species collected in Quesnel Lake, results were similar to largescale sucker in that condition factors were generally consistent both between years and between regions in the lake (Figure 40). The only notable difference was that condition factor was higher for northern pikeminnow in 2015 compared to 2016. However, in 2015 pikeminnow were only collected in the West Basin, and therefore a regional comparison for that year could not be conducted. The difference in condition factors between 2015 and 2016 for the West Basin may have been due to differences in sample methodology. Northern pikeminnow were caught using trap nets in 2015 whereas they were caught using hoop nets in 2016, which resulted in older and overall larger fish being caught in 2016.

[48] For all species other than sockeye salmon, comparisons were made between fish collected in the West Basin (referring to the area within the West Arm that is west of the sill near Caribou Island) compared to other regions of the lake. Comparisons for sockeye salmon were made between fish collected in the West Arm compared to other regions of the lake (rather than the West Basin specifically) due to lack of additional sampling information.

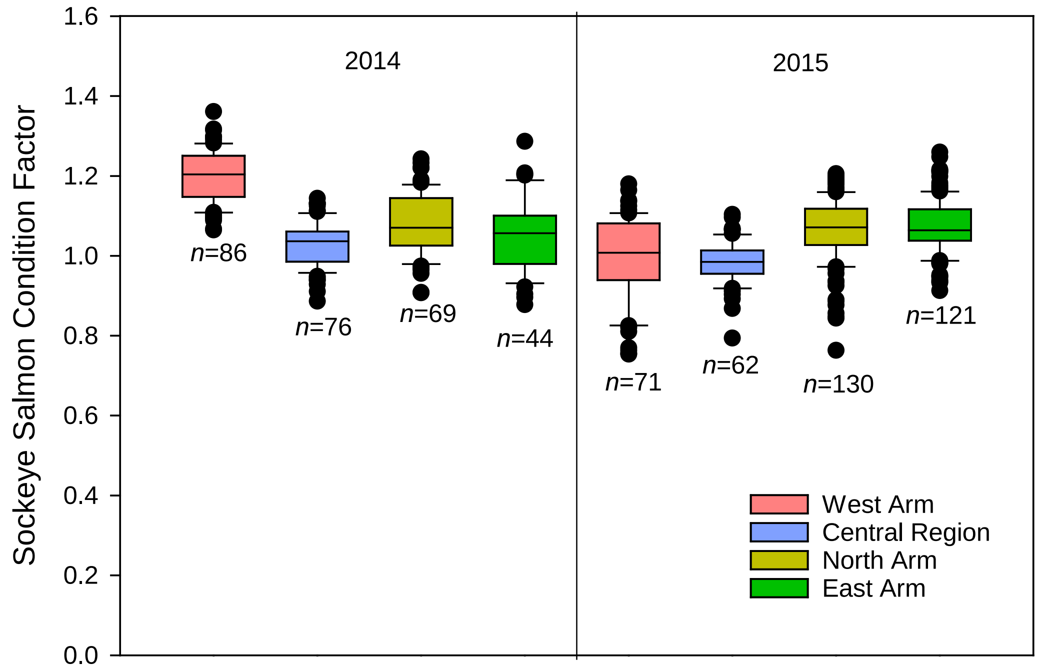

Figure 38: Condition Factors for Juvenile Sockeye Salmon Collected from Four Regions of Quesnel Lake, 2014 and 2015

Note: Boxplots indicate minimum values, 25th, 50th, and 75th percentiles, maximum values, and outliers. The West Arm is considered exposed area and the Central Region, North Arm, and East Arm are considered reference areas.

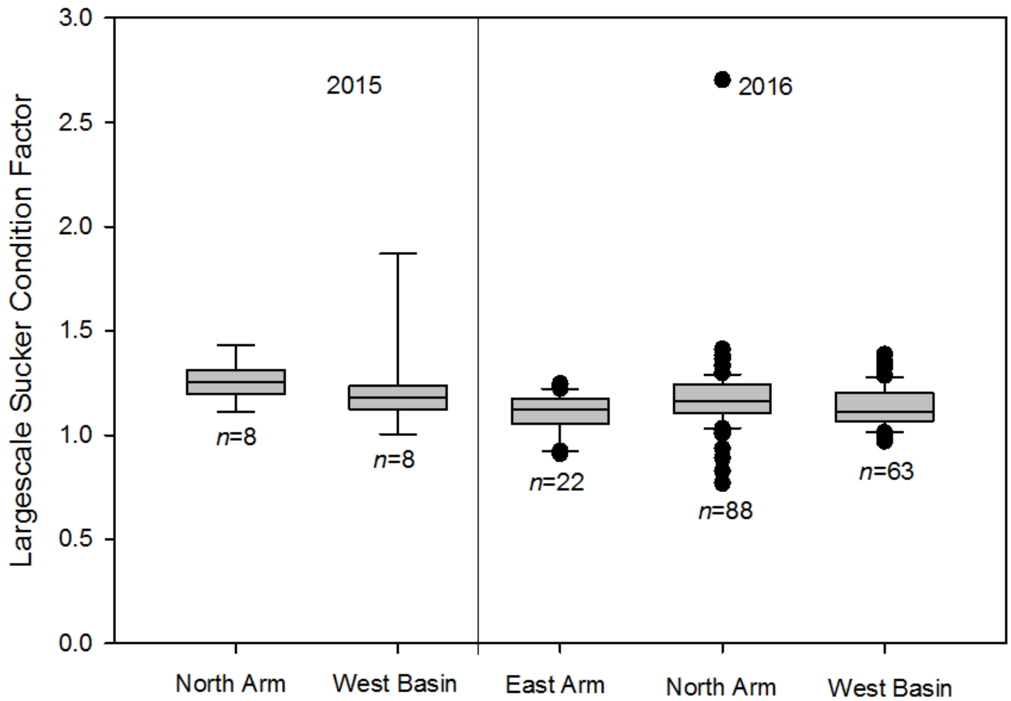

Figure 39: Condition Factor for Largescale Sucker Collected from Quesnel Lake Exposed (West Basin) and Reference Areas (North Arm and East Arm), 2015 and 2016

Note: Boxplots indicate minimum values, 25th, 50th, and 75th percentiles, maximum values, and outliers.

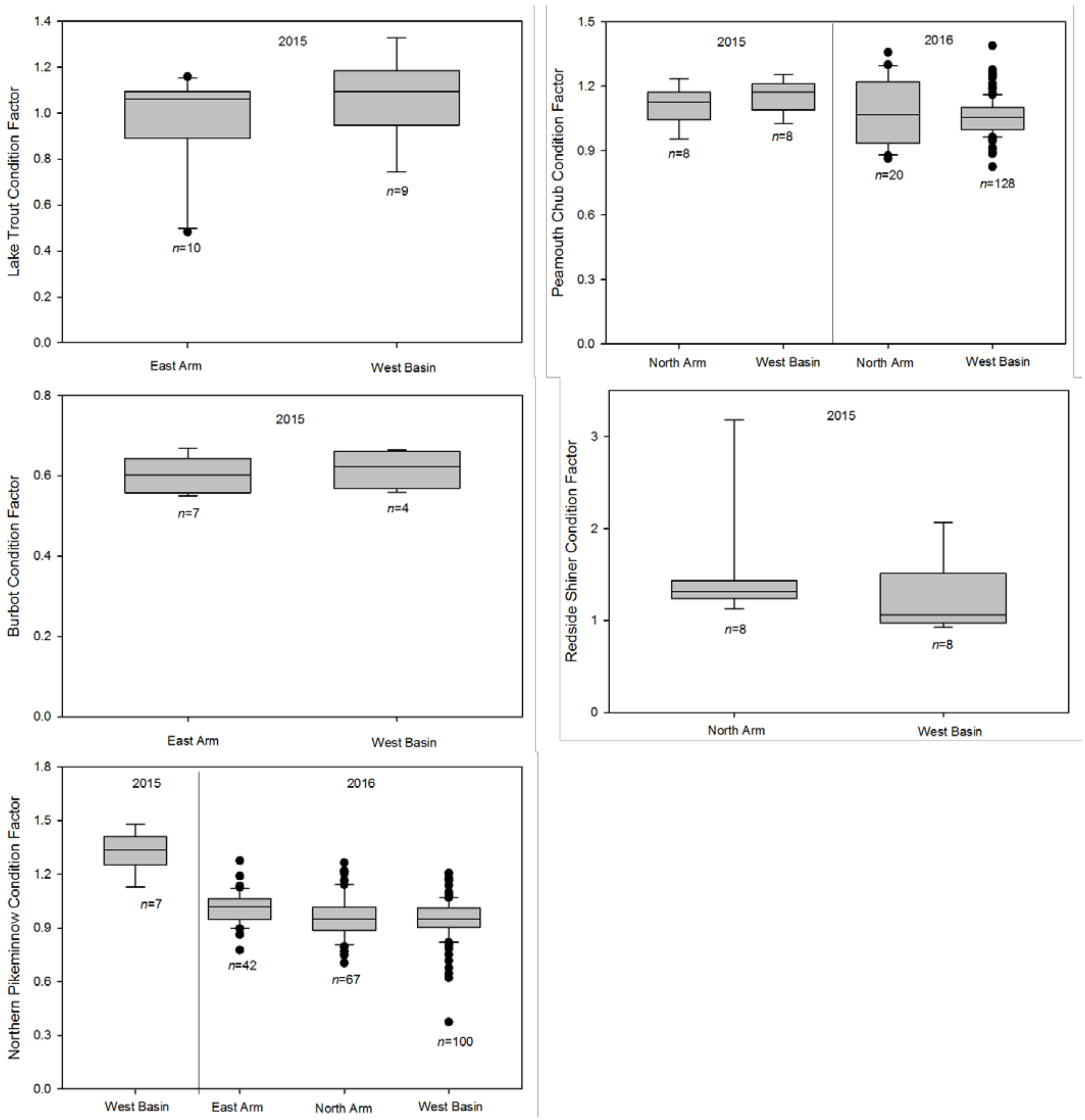

Figure 40: Condition Factors for Lake Trout, Peamouth Chub, Burbot, Redside Shiner, and Northern Pikeminnow Collected from Quesnel Lake Exposed (West Basin) and Reference Areas (North Arm and East Arm), 2015 and 2016

Note: Boxplots indicate minimum values, 25th, 50th, and 75th percentiles, maximum values, and outliers.

4.5.5.3 Edney Creek

As noted in Section 2.4.1, MPMC has been implementing a rehabilitation strategy over the last two years that resulted in the removal of tailings from the Hazeltine Channel bed and surrounding areas, reconstruction of the Hazeltine and Edney Creek channels, and restoration of the connection of Edney Creek with Quesnel Lake. The lower area of Edney Creek was reconstructed on a prioritized basis to stop erosion and restore fish access (completed February 2015), with instream habitat features subsequently added (initial work August 2015). Fish access to Hazeltine Creek is still restricted at this time due to ongoing rehabilitation activities, but access to Edney Creek was restored in February 2015. Throughout 2016, all measured parameters in Edney Creek were below WQGs or within background concentrations.

Twelve fish species were captured in 2016 from Edney Creek, including juvenile rainbow trout, and coho and chinook salmon, confirming that the connection between Edney Creek and Quesnel Lake is functioning. Other species observed were: Kokanee salmon, mountain whitefish, northern pikeminnow, burbot, largescale sucker, longnose sucker, bridgelip sucker, longnose dace, peamouth chub, and redside shiner. Rainbow trout and longnose dace were the most abundant and the only fish with sufficient numbers for comparison between the reconstructed habitat and the reference location. Mean condition factors were similar between areas of Edney Creek for rainbow trout (Figure 41) and longnose dace (Figure 42). It should be noted that the fish collected from the lower, reconstructed area of the creek cannot be considered a separate population from fish collected in the upstream reference area because there are no physical barriers that prevent the movement of fish between areas.

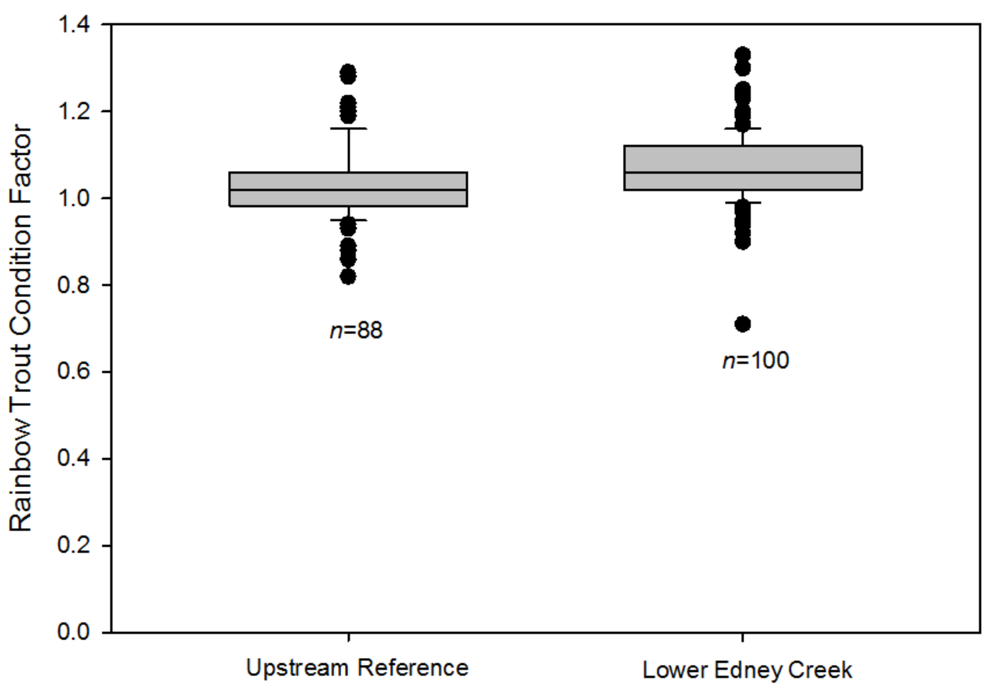

Figure 41: Condition Factors for Rainbow Trout Collected from Edney Creek Exposed (Lower Edney Creek) and Reference Areas (Upstream Reference), 2016

Note: Boxplots indicate minimum values, 25th, 50th, and 75th percentiles, maximum values, and outliers.

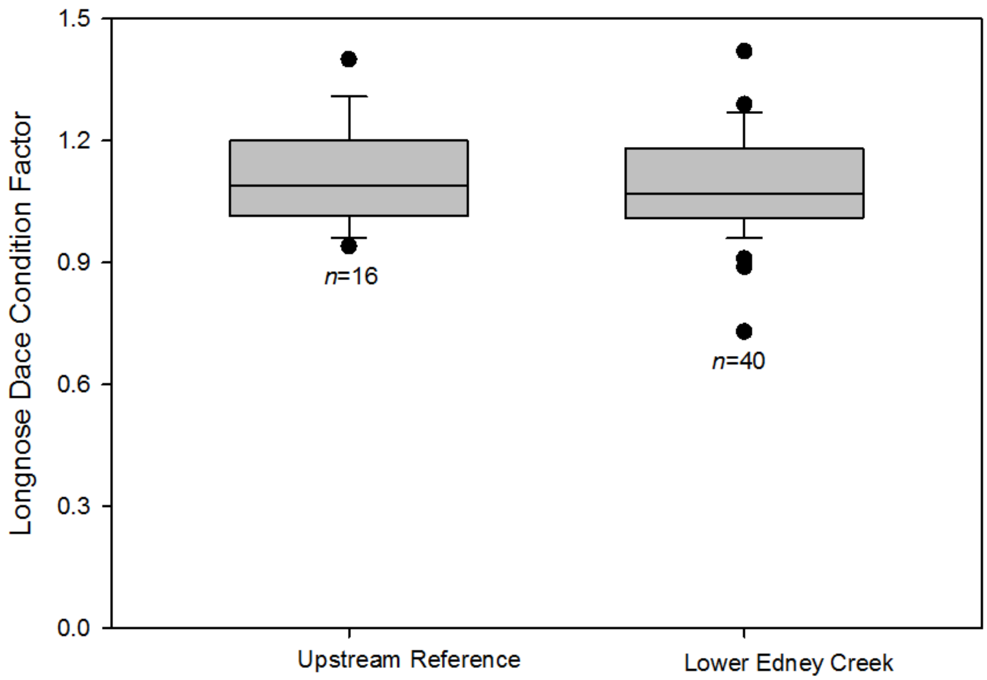

Figure 42: Condition Factors for Longnose Dace Collected from Edney Creek Exposed (Lower Edney Creek) and Reference Areas (Upstream Reference), 2016

Note: Boxplots indicate minimum values, 25th, 50th, and 75th percentiles, maximum values, and outliers.

4.5.5.4 Comparison to Decision Criteria

A specific numerical decision criterion was not developed for fish condition factor because it is difficult to quantify a difference in condition factor that would indicate an adverse impact on fish health. Condition factor is a ratio between length and weight that serves as a measure of overall body condition and is intended to be evaluated in conjunction with other lines of evidence concerning fish in order to more comprehensively characterize the status of fish health and habitat quality. Condition factors were qualitatively compared between available years for fish in Polley Lake and between regions and available years for fish in Quesnel Lake to evaluate whether or not condition factor was reduced in areas expected to be affected by the TSF embankment breach relative to reference areas. Data for juvenile sockeye salmon were given highest priority in Quesnel Lake due to the large sample sizes, with other fish species providing additional contextual information and support. The available data on fish condition indicates that condition factors were similar between years in Polley Lake and between years and regions in Quesnel Lake for all species considered.

Uncertainty for this line of evidence is primarily attributed to confounding factors and sources of variation in the sampled populations. Fish growth and foraging behaviour can be influenced by a variety of factors beyond contaminant exposure, including interspecies competition and variations in population density. In addition, data available for some species collected in Quesnel Lake represented a subsample of fish collected that were retained for tissue chemistry analysis, and selection methods for tissue analysis may have influenced the results. Variations in sampling methodology between years also contributes to uncertainty in this line of evidence. This line of evidence should be considered in combination with other lines of evidence relating to fish health and habitat quality. Where a difference is shown between impacted and background conditions, condition factor is a function of fish weight and length, which is directly related to the growth endpoint. Therefore, the strength of association to the assessment endpoint is considered moderate.

LOE	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Fish Condition	Protect fish from reductions in survival, growth and reproduction as a result of multiple exposure pathways.	Assess the condition factor of rainbow trout and longnose suckers in Polley Lake.	Average condition factor measured post-breach is not reduced relative to pre-breach.	Polley Lake	Meets decision criterion
		Assess the condition factor of juvenile sockeye salmon collected from different sampling zones in Quesnel Lake.	Average condition factor among sampling zones within Quesnel Lake are similar.	Quesnel Lake	Meets decision criterion
		Assess the condition factor of rainbow trout and longnose dace in Edney Creek	Average condition factor among sampling zones within Edney Creek are similar.	Edney Creek	Meets decision criterion

4.5.6 Fish Age Frequency Distribution

Age frequency distributions are commonly used to evaluate community structure in fisheries assessments. Determination of age frequency may be influenced significantly by sampling methodology due to size selectivity of nets and age and size-dependent habitat and feeding preferences.

4.5.6.1 Study Design and Methodology

Information on fish collected from Polley Lake and Quesnel Lake is provided in Appendix P-2 and P-3. In brief, sampling data were available for both rainbow trout and longnose sucker in Polley Lake from 2014 (immediately after the TSF embankment breach) and 2016. In both sampling events, rainbow trout and longnose sucker were collected via gillnets (38 to 89 mm mesh size) that were placed in similar locations in the lake for similar durations of time. Fish were collected as part of a survey that included determination of sex, length and weight measurements, and observations of parasites. A subsample of the fish collected (83 rainbow trout and 78 longnose sucker in 2014 and 45 rainbow trout and 29 longnose sucker in 2016) were retained and sacrificed for aging via otolith analysis. Results of age analysis were compiled for each species and fish were grouped by year. The number of each fish of each age was converted to a proportion of total fish aged for each species and age frequency distributions were compared between years to investigate potential changes in community structure that may have been influenced by the breach.

4.5.6.2 Summary of Findings

The age frequency distribution for rainbow trout in 2016 was shifted to the right compared to the distribution for 2014, with an apparent absence of rainbow trout in age groups of 1 and 2 years old in 2016 (Figure 43). This indicates a likely reduction in recruitment in 2014 and 2015, following the TSF embankment breach. The age frequency distributions for longnose sucker in 2014 and 2016 were more similar, although in 2016 none of the aged longnose sucker belonged to the 2 year old age class, indicating further evidence for a potential impact of the breach (Figure 44).

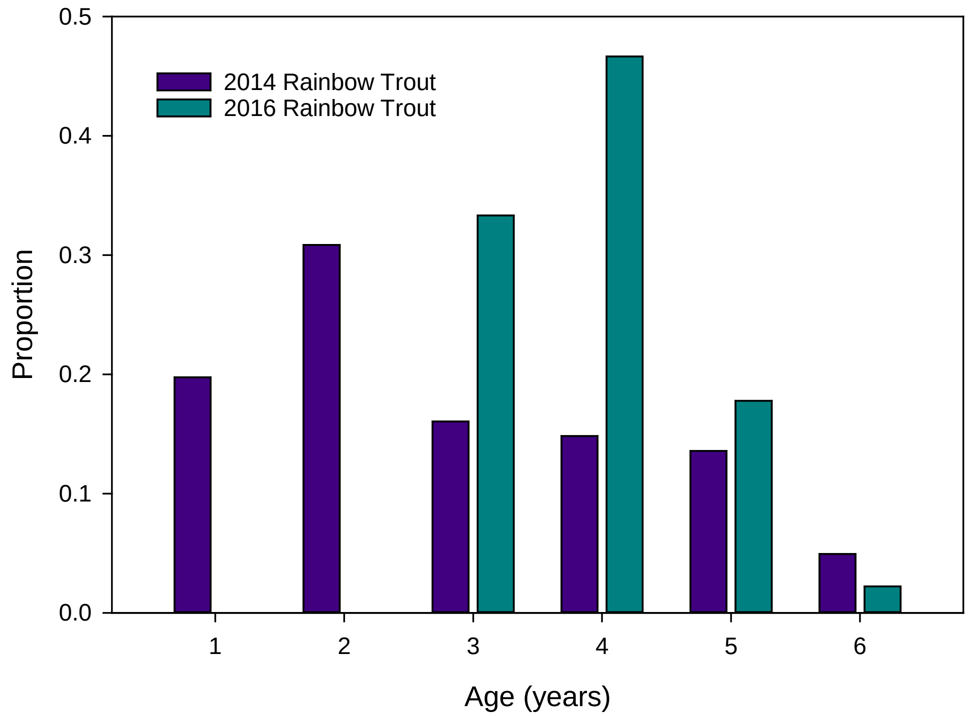

Figure 43: Age Frequency Distributions for Rainbow Trout Collected From Polley Lake in 2014 and 2016

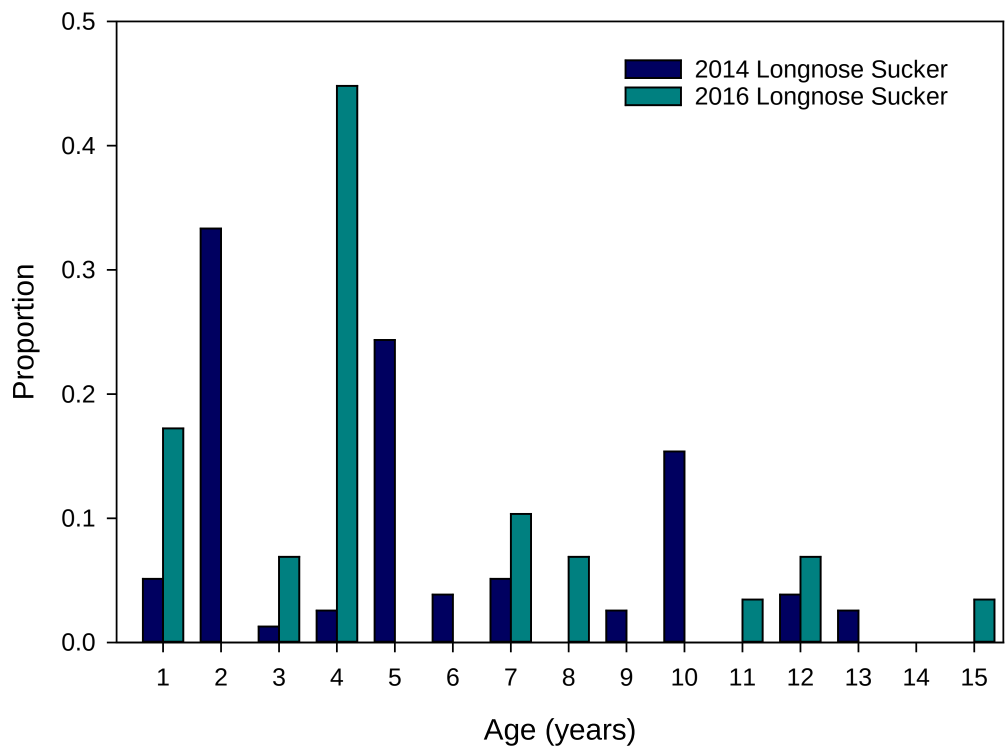

Figure 44: Age Frequency Distributions for Longnose Sucker Collected From Polley Lake in 2014 and 2016

As noted by Minnow (Appendix P-3), longnose sucker spawn in the spring and early summer in streams and shallow areas of the lake, and the young-of-year typically leave spawning areas by late fall. Similarly, rainbow trout tend to spawn in spring and hatch 1 to 2 months later. Following hatching, rainbow trout will take approximately two weeks to reabsorb their yolk sacs, after which they may leave spawning areas or may remain for an extended period of time. The influx of tailings material in August 2014 may have affected the young-of-year of both species in 2014, as well as reducing spawning habitat size and quality (Appendix P-4). In addition, shortly after the TSF breach a fish fence was installed at the outlet of Polley Lake that feeds into the Hazeltine Channel. This fish fence has prevented rainbow trout in Polley Lake from accessing historical spawning habitat in the Hazeltine Channel during ongoing reconstruction and remediation of the Hazeltine Channel. The reduced availability of spawning habitat as a result of the temporary fish fence may also affect recruitment post-breach.

4.5.6.3 Comparison to Decision Criteria

A numerical decision criterion was not developed for fish age structure because a meaningful quantitative difference in age frequency distribution is difficult to operationalize. The age frequency information is intended to be considered in conjunction with other lines of evidence concerning fish to more comprehensively characterize the status of fish health and habitat quality. The decision criterion originally proposed by Golder was similar age frequency distributions between fish caught in 2014 and those caught historically prior to the breach. However, insufficient historical fish data that includes age distributions were available for this comparison.

Therefore, the age frequency distributions for longnose sucker and rainbow trout in Polley Lake were qualitatively compared between 2014 and 2016 to identify whether there has been a change in recruitment of young fish between years. Based on the available data, the age frequency information indicates a reduction in recruitment for rainbow trout following the TSF embankment breach, as well as a potential loss of the majority of the young-of-year for both rainbow trout and longnose sucker in 2014.

Uncertainty is primarily attributed to confounding factors and sources of variation in the sampled fish populations. Age distributions may be influenced by to a number of factors that affect spawning and recruitment success that are unrelated to the breach. In addition, measured age frequency may be skewed by factors such as size selectivity of mesh openings in gillnets or age-specific habitat preferences and behaviours. This line of evidence should be considered in combination with other lines of evidence relating to fish health and habitat quality as a change in the age distribution could occur from a combination of direct effects (i.e., toxicity) or indirect effects (e.g., reduced food availability; loss of habitat) to the fish population.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Fish Age Frequency	Protect fish from reductions in survival, growth and reproduction as a result of multiple exposure pathways.	Assess the age frequencies of rainbow trout and longnose suckers in Polley Lake.	No change in recruitment of young fish between years based on post-breach age frequency distributions	Polley Lake	Does not meet decision criterion

4.6 Exposure-Based LOE: Sediment Chemistry

4.6.1 Sediment Chemistry

The following parameters were identified as COPCs in sediment based on chemistry data collected in 2014 and 2015, as described in the aquatic problem formulation.

- Copper and arsenic were identified as COPCs because 90th percentile concentrations exceeded the CSR Schedule 9 sensitive sediment criterion ($SedQC_{SS}$) and background for Polley Lake, the Hazeltine Channel, and Quesnel Lake.
- Sediment quality was not assessed in Quesnel River because this habitat is erosional and no suitable areas of sediment deposition were identified during sampling.
- Only the COPCs copper and arsenic are discussed in the summary of sediment quality findings for the sediment chemistry line of evidence below.

4.6.1.1 Study Design and Methodology

The assessment of sediment chemistry relied on sediment data collected by Minnow in 2014, 2015, and 2016. The 2014 and 2015 sediment sampling focused on initial characterization of the potential impacts from the TSF embankment breach on receiving environment sediment quality[49] as well as sediment traps placed in 2014 and 2015. The 2016 sampling was conducted to verify sediment and porewater chemistry at previously sampled locations, to further map the spatial extent of impact of the TSF embankment breach, and to provide supporting data for a sediment toxicity investigation. The 2016 sampling locations and sample collection methodology are summarized below. Details are provided in Appendix A-7.2[50].

- Sediment samples were collected at shallow, mid-depth, and deep locations *(n*=5 stations/area) for both exposed and reference locations in Bootjack Lake (reference), Polley Lake (mid-depth and deep, exposed), and Quesnel Lake (shallow/littoral and deep/profundal, exposed and reference). In Polley Lake, sediments were collected separately from the north and south locations in the lakebed, and in Quesnel Lake, sediment samples were collected separately from near-field and far-field locations.
- Samples were also collected in lower Hazeltine Channel[51] *(n*=5 replicates), which receives eroded material (both tailings impacted and native) from exposed areas along the creek. Spatial, in-fill samples were collected at stations in Polley Lake (*n*=3) and in Quesnel Lake (*n*=14 deep/profundal). In addition, samples were collected from 10 deep exposed "intermediate" locations in Quesnel Lake (*n*=1 sample/location) that had been selected to represent a range of physical and chemical characteristics to evaluate the influence of sediment characteristics on sediment toxicity (see Section 4.7.1).

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[49] The 2014 dataset consisted of 75 stations from exposed areas of the Hazeltine Channel, Polley Lake, and Quesnel Lake and 31 stations from reference areas in Bootjack Lake and Quesnel Lake. The 2015 data set consisted of 25 exposed stations from the Hazeltine Channel, Polley Lake, and Quesnel Lake and 13 reference stations in Bootjack Lake and Quesnel Lake.

[50] See Figures 1 and 2 and Tables A.1 and A.2 of Appendix A-7.2 for sediment sampling locations, including the infill and sediment toxicity locations sampled in 2016.

[51] Hazeltine Channel sediment samples were collected from the Creek in 2014, and from the sedimentation pond in 2015 and 2016. Concentrations were generally similar across the years despite the relocation of monitoring stations.

- Samples were typically collected using a Kajak-Brinkhurst or Tech Ops corer for locations in Polley Lake and Bootjack Lake, and either a petite or standard stainless steel ponar grab sampler for locations in Quesnel Lake and the Hazeltine Channel sedimentation pond, consistent with previous sampling methods where possible. Composite samples were collected from the top 3 cm of sediment retrieved from a designated minimum number of acceptable grabs.

- Samples were analysed for metals, total organic carbon (TOC), total nitrogen, total sulphur, percent moisture, pH, and particle size. Metals, pH, and TOC were measured on the silt/clay (<63 µm) fraction of sediment, whereas percent moisture, pH, total nitrogen, total sulphur, and grain size were measured on bulk sediment. In 2014, metals were also analysed in the <2 mm fraction of sediment (i.e., sand and fines), and results indicated similar concentrations for both fractions of sediment[52].

- Minnow provided a summary of all sediment chemistry and reported mean sediment concentrations for each year (2014, 2015, 2016) for the different sampling locations in each waterbody, as well as updated estimates of reference concentrations[53]. Mean sediment concentrations were compared against CSR Schedule 9 "sensitive" ($SedQC_{SS}$) and "typical" ($SedQC_{TS}$) sediment standards, as well as reference concentrations.

- Sediment traps were placed in Quesnel Lake and Polley lake in two rounds: a "winter" set from August 2014 (Quesnel Lake) or October 2014 (Polley Lake) to May 2015 and a "summer" set from May 2015 to August 2015. There were three stations in Quesnel Lake including; one near-field station at the mouth of Hazeltine Creek, one far-far-field station near Cedar Point Park, and one reference station in Horsefly Bay. There were two stations in Polley lake located at sampling stations P1 (north basin) and P2 (south basin). Sediment collected from the traps were used to estimate the deposition rate of sediment in millimetres per year (mm/yr) and bulk sediment chemistry (Appendix K-5). Sediment from the traps was also analyzed using sequential extraction (See Section 0)

4.6.1.2 Potentially Confounding Factors

Particle size distributions have the potential to influence the partitioning and distribution of sediment-associated contaminants. Laboratory analyses for metals in sediment were performed on the fine (silt-clay) portion of the sediment sample only (<63 µm fraction). This is consistent with the BC MoE *Water and Air Baseline Monitoring Guidance Document for Mine Proponents and Operators*, which states that "finer sediment particles are of greater interest in terms of contaminant loads, because most mine-related chemical contaminants preferentially bind to silts and clays" (BC MoE 2012a). The particle size distribution data for tailings samples collected by SNC-Lavalin in Hazeltine Channel in 2014 indicated that tailings samples were mostly sand and silt[54]. Minnow reported that 2014 sediment chemistry results were similar in the fine and coarser portions (<63 µm and <2 mm, respectively)[55]. Therefore, particle size distribution should not significantly affect the sediment chemistry results, even for variable bulk sediment composition.

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[52] Refer to Figures 5.1, 6.1, 7.1, and 8.1 of MPMC 2015 Appendix E for comparison of chemistry on <2 mm and <63 µm fractions.

[53] The highest 95th percentile value calculated for each of the historic, 2014, 2015, or 2016 reference location datasets was selected as the reference concentration for each waterbody.

[54] MPMC 2015 Appendix D - Soil Quality Impact Assessment prepared by SNC-Lavalin (refer to Table M)

[55] Refer to Figure 5.5 in MPMC 2015 Appendix E

Other physical characteristics such as TOC can also influence the interpretation of chemistry data; however, the CSR Schedule 9 criteria do not consider TOC normalization, and thus this influence has not been evaluated. Physical characteristics of sediment do have the potential to be confounding factors in the interpretation of biological effects data and spatial variation in sediment physical characteristics are discussed further in Section 4.6.5.

4.6.1.3 Summary of Findings

The following discussion of sediment chemistry results focuses on the COPCs identified in sediment – arsenic and copper. Table 36 and Table 37 summarize concentrations of these metals in each waterbody.

Table 36: Summary of 2014-2016 Copper Concentrations (mg/kg dw)

Area	n	Minimum	Average	Standard Deviation	Maximum	Reference[a]
Polley Lake	44	69	**<u>611</u>**	158	**<u>864</u>**	**<u>406–510</u>**
Hazeltine Channel	25	**152**	**<u>434</u>**	202	**<u>851</u>**	95
Quesnel Lake littoral	40	26	**<u>330</u>**	294	**<u>1020</u>**	49
Quesnel Lake profundal	91	41	**<u>470</u>**	325	**<u>1260</u>**	61

CSR Sensitive Sites standard = 120 mg/kg, CSR Typical Sites standard = 240 mg/kg

Bold concentrations in exceed the $SedQC_{SS}$, **<u>Bold underlined</u>** concentrations also exceed the $SedQC_{TS}$.

Quesnel Lake profundal summary statistics include intermediate/infill samples collected in Quesnel Lake.

(a) Reference conditions are 95th percentiles presented in Appendix A-7.2. Range for Polley Lake represents deep and mid-depth samples, respectively, and are predominantly based on historical bulk sediment concentrations.

Table 37: Summary of 2014–2016 Arsenic Concentrations (mg/kg dw)

Area	n	Minimum	Average	Standard Deviation	Maximum	Reference[a]
Polley Lake	44	5.3	**12**	1.8	**15**	8.9–**13**
Hazeltine Channel	25	7.3	**13**	3.2	**17**	**12**
Quesnel Lake littoral	40	1.8	9.3	5.1	**19**	5.3
Quesnel Lake profundal	91	4.5	**<u>23</u>**	24	**<u>165</u>**	**<u>21</u>**

CSR Sensitive Sites standard = 11 mg/kg, CSR Typical Sites standard = 20 mg/kg

Bold concentrations in exceed the $SedQC_{SS}$, **<u>Bold underlined</u>** concentrations also exceed the $SedQC_{TS}$.

(a) Reference conditions are 95th percentiles presented in Appendix A-7.2. Polley Lake range bounds deep and mid-depth samples.

Quesnel Lake profundal summary statistics include intermediate/infill samples collected in Quesnel Lake.

The key findings from Table 36 and Table 37 were:

- For Polley Lake, concentrations were generally consistent among sampling years 2014, 2015, and 2016. The mean concentrations of copper and arsenic were similar between the north and south areas of Polley Lake. Mean copper concentrations were approximately 5-times higher than the CSR Schedule 9 $SedQC_{SS}$ and 2.5-times the $SedQC_{TS}$, and up to 1.5-times higher than reference concentrations for all sampling years in all areas of the lake. Copper concentrations in reference location sediments were also above both the $SedQC_{SS}$ and $SedQC_{TS}$. Mean arsenic concentrations in Polley Lake marginally exceeded the $SedQC_{SS}$ and were within the range of reference concentrations for all sampling years and all sampling areas. Concentrations of arsenic did not exceeded the $SedQC_{TS}$.
- For the Hazeltine Channel, data were limited to samples were collected from lower Hazeltine Channel in 2014 and from the Hazeltine sedimentation pond in 2015 and 2016. The Hazeltine Channel was remediated and tailings were removed throughout 2015 and 2016, which did not allow for collection of samples in the channel during this time. The sedimentation pond is a man-made maintained watercourse designed to collect suspended material prior to discharge to Quesnel Lake and is not currently a fish habitat (Appendix A-7.1). Mean copper concentrations for all sampling years were up to 3.6-times higher than the $SedQC_{SS}$, 1.8-times the $SedQC_{TS}$, and 4.6-times the reference concentration. Mean arsenic concentrations marginally exceeded the $SedQC_{SS}$, but did not exceed the $SedQC_{TS}$, and were similar to the reference concentration. Copper concentrations have remained similar between 2014 and 2016 in the sedimentation ponds (i.e., 3.3 to 3.9-times the $SedQC_{SS}$).
- For the littoral areas of Quensel Lake, mean concentrations of copper and arsenic were generally similar through 2014–2016 and were typically lower in the far-field area than in the near-field, with no exceedances of applicable standards for mean concentrations in the far-field area throughout 2014–2016. Mean arsenic and copper concentrations in the near-field area exceeded the $SedQC_{SS}$ as well as reference concentrations throughout 2014–2016. In 2016, the mean copper concentration was 5.5-times the $SedQC_{SS}$, 2.7-times the $SedQC_{TS}$, and 13-times the reference concentration (Table 6 of Appendix A-7.2). The mean arsenic concentration in 2016 was 1.3-times the $SedQC_{SS}$ and 2.8-times the reference concentration (Table 6 of Appendix A-7.2). Concentrations of arsenic did not exceeded the $SedQC_{TS}$.
- For the profundal areas of Quesnel Lake, mean arsenic and copper concentrations exceeded the $SedQC_{SS}$, in both the near-field and far-field exposed areas throughout 2014–2016. Mean copper concentrations also exceeded the $SedQC_{TS}$ and reference concentrations. Mean arsenic concentrations were below the $SedQC_{TS}$ and reference concentrations (Table 4 of Appendix A-7.2). In 2016, the mean copper concentration in the near-field area was slightly higher than in previous years, and is now approximately 5-times the $SedQC_{TS}$. The copper concentrations in the far-field area was 2.4-times the $SedQC_{TS}$, similar to 2014[56]. The mean arsenic concentration in the near-field area in 2016 was 1.7-times the $SedQC_{SS}$, which is a small increase relative to 2014 and 2015. The arsenic concentration in the far-field area was 1.2-times the $SedQC_{SS}$ (Table 4 of Appendix A-7.2). Overall, concentrations of copper in profundal sediments in Quesnel Lake appear to have increased between 2014 and 2016 (by 10 and 50%, respectively), and concentrations remain greater in the near-field than far-field areas.

[56] The profundal far-field area was not sampled in 2015.

Table 38: Summary of 2014 and 2015 Sediment Trap Copper Concentrations (mg/kg dw)

Area	n		Average		Maximum		Reference
	2014	2015	2014	2015	2014	2015	
Polley Lake north basin (P1)	4	6	**669**	**577**	**727**	**630**	NA
Polley Lake south basin (P2)	6	6	**737**	**540**	**799**	**574**	NA
Quesnel Lake near-field	5	5	**1144**	**239**	**1160**	**376**	72.1
Quesnel Lake far-field	3	5	**238**	**317**	**300**	**738**	

CSR Sensitive Sites standard = 120 mg/kg, CSR Typical Sites standard = 240 mg/kg.

Bold concentrations in exceed the SedQC$_{SS}$, **Bold underlined** concentrations also exceed the SedQC$_{TS}$.

Quesnel Lake profundal summary statistics include intermediate/infill samples collected in Quesnel Lake.

The reference sediment for Quesnel Lake represents the 95th percentile of reference sediments collected from Horsefly Bay.

Table 39: Summary of 2014 and 2015 Sediment Trap Arsenic Concentrations (mg/kg dw)

Area	n		Average		Maximum		Reference
	2014	2015	2014	2015	2014	2015	
Polley Lake north basin (P1)	4	6	14.5	12.8	15.1	14.3	NA
Polley Lake south basin (P2)	6	6	15.3	12.9	17.0	17.2	NA
Quesnel Lake near-field	5	5	17.0	8.77	17.2	15.0	8.91
Quesnel Lake far-field	3	5	92.4	82.9	98.6	108	

CSR Sensitive Sites standard = 11 mg/kg, CSR Typical Sites standard = 20 mg/kg

Bold concentrations in exceed the SedQC$_{SS}$, **Bold underlined** concentrations also exceed the SedQC$_{TS}$.

The reference sediment for Quesnel Lake represents the 95th percentile of reference sediments collected from Horsefly Bay

The key findings from the Quesnel Lake and Polley Lake sediment traps were (Table 38 and Table 39):

- For Quesnel Lake, the near-field mean copper concentration was approximately 9.5-times the CSR Schedule 9 SedQC$_{SS}$ and 4.8-times the SedQC$_{TS}$ in 2014, however, mean copper concentrations were lower in 2015 exceeding the SedQC$_{SS}$ by 2.0-times and marginally below the SedQC$_{TS}$ (Table 38). At the near-field station mean arsenic concentrations were approximately 1.5-times the SedQC$_{SS}$ but below the SedQC$_{TS}$ in 2014, and below both standards in 2015 samples (Table 39). Concentrations of metals at the far-far-field station did not have a distribution typical to areas impacted by the TSF embankment breach with higher concentrations of arsenic and lower concentrations of copper (Appendix K-1; Table 38 and Table 39). Mean concentrations of copper and arsenic were higher than reference for both the near-field and far-far-field stations.
- For Quesnel Lake, it was estimated that it would take approximately four years (far-far-field) to eight years (near-field) for 1 cm of sediment to deposit. Deposition rates at the reference station were highly variable and much higher with an estimated 0.5 to 2.4 cm of sediment deposited yearly.
- For Polley Lake, it was estimated that it would take approximately one to three years for 1 cm of sediment to deposit with higher rates measured in the south basin (P2) than the north basin (P1). Mean copper concentrations were approximately 4.5-times to 6.1-times higher than CSR Schedule 9 SedQC$_{SS}$ and 2.3-times to 3.1-times higher than the SedQC$_{TS}$ at stations P1 and P2 (Table 38). Mean arsenic concentrations were approximately 1.2-times to 1.4-times higher than CSR Schedule 9 SedQC$_{SS}$ but lower than the SedQC$_{TS}$ in samples at stations P1 and P2. Concentrations of both copper and arsenic were lower in 2015 samples than 2014 samples (Table 38 and Table 39).

4.6.1.4 Quesnel Lake Spatial Summary

The spatial extent of copper and arsenic concentrations in Quesnel Lake was evaluated using GIS. The purpose of the analysis was to determine the 2-D gradient of concentrations. Concentrations between sampling locations were interpolated using an iterative finite difference interpolation technique to create a smoothed surface. The surface was separated into different bands based on the CSR Schedule 9 sensitive standard ($SedQC_{SS}$) for copper or arsenic, 2-times $SedQC_{SS}$ and 10-times $SedQC_{SS}$.

Figure 45 and Figure 46 show the concentration gradients for copper and arsenic in littoral and profundal sediments in the West Basin of Quesnel Lake. Overall, these figures show that:

- Copper concentrations were consistently lower in littoral sediment than profundal sediment. This difference was sufficiently pronounced as to justify using bathymetry as a primary factor in the extrapolation. Sediment collection in the littoral zone was primarily along the eastern side[57]. This resulted in reduced sample coverage in some littoral areas which hindered interpolation of concentration gradients (i.e., profundal samples were not bounded by a littoral sample closer to shore). Copper concentration data from the littoral stations were reviewed, and with the exception of the near-field area at the mouth of the Hazeltine Channel, it was inferred that copper concentrations in the most of the littoral zone would be expected to be below the $SedQC_{SS}$.
- Copper concentrations in sediment generally decrease with distance from the mouth of the Hazeltine Channel, with the highest concentrations in the near-field profundal zone downstream of the Hazeltine Channel. This concentration pattern is consistent with the understanding of the debris flow and deposition of tailings-impacted materials into Quesnel Lake following the breach.
- Arsenic concentrations in sediment of the West Basin followed a different pattern than copper. The highest arsenic concentrations were observed in the far-field east and west locations, rather than near the mouth of the Hazeltine Channel (Figure 46). This suggests that the distribution of arsenic in sediment is not related to the breach, but likely due to natural sources of variation in the spatial distribution of arsenic.

The surface area (km^2) of each concentration band for copper was calculated and is summarized in Table 38. The proportion of the total area of Quensel Lake that was part of the original PEEIAR assessment (i.e., the West Basin) that has substantial exceedances of the numerical standards is less than 10%.

[57] In 2016, four of the 21 selected in-fill locations could not be sampled due to inappropriate substrate type, despite multiple sampling attempts at the sampling location and in the immediately surrounding area. Two of these stations were along the western side of the West Basin and the other two stations were near the mouth of the Hazeltine Channel (see Table A.2 of Appendix A-7.2).

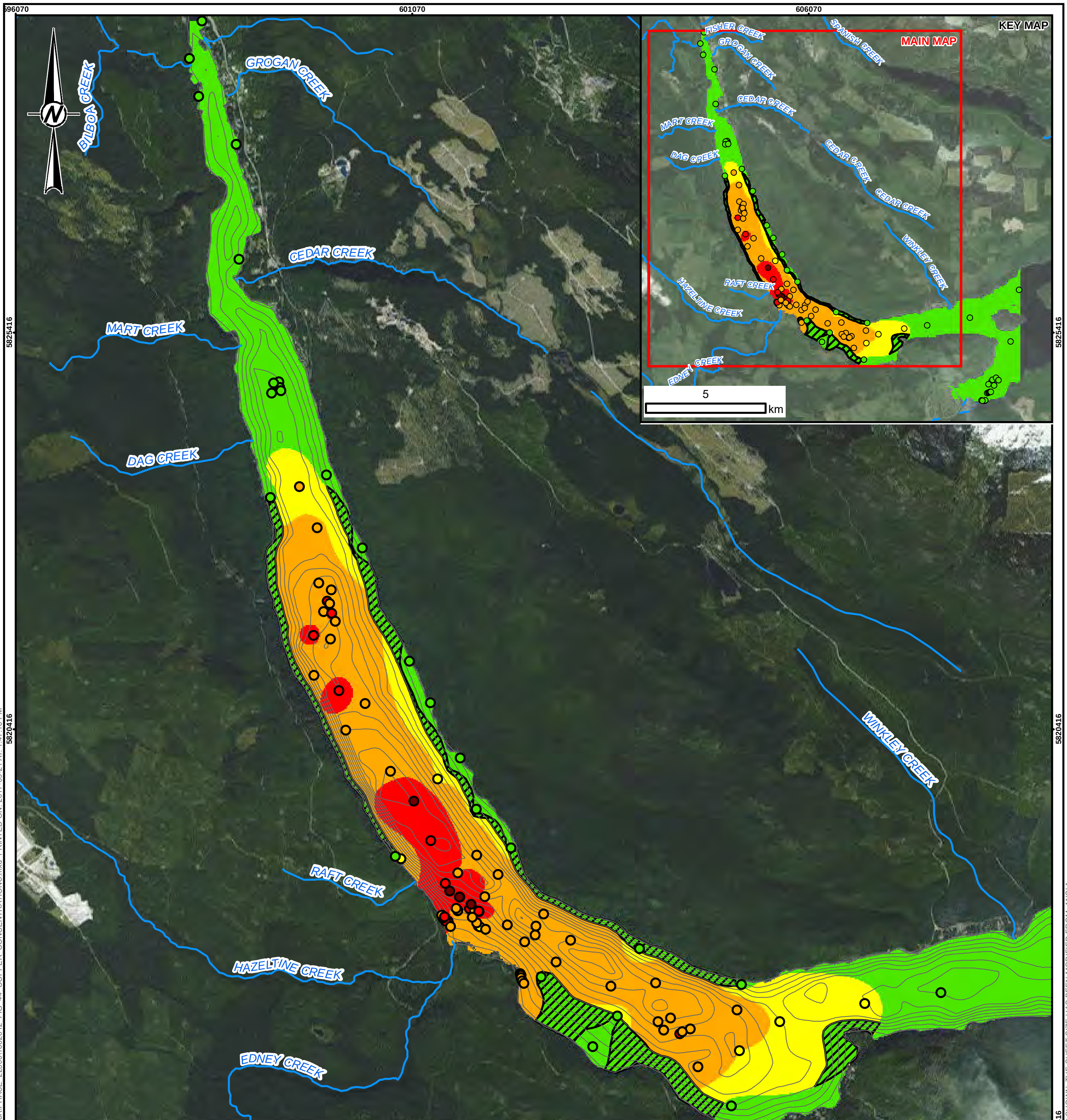
Table 40: Area of the Quesnel Lake West Basin Associated with Exceedance of the CSR Sediment Standards for Copper

Band[a]	Copper Concentration (mg/kg dw)	3D Bathymetric Surface (km^2)	% of West Basin Area[b]
Green	<120 (<$SedQC_{SS}$)	10.8	46%
Yellow	120 – 240 (<2x $SedQC_{SS}$; <$SedQC_{TS}$)	3.08	13%
Orange	240 – 600 (<5x $SedQC_{SS}$)	8.36	35%
Red	600 – 1,200 (<10x $SedQC_{SS}$)	1.41	6%
Dark Red	≥1,200 (≥10x $SedQC_{SS}$)	0.00141	0.006%

$SedQC_{SS}$ = CSR sediment quality standard for sensitive sites of 120 mg/kg dw; $SedQC_{TS}$ = standard for typical sites of 240 mg/kg dw.

(a) The colour progression indicates increasing exposure concentrations only and is not intended to reflect potential for adverse ecological conditions.

(b) For the purposed of calculating the total area, the West Basin was defined to include the area west of the sill and the west-most side of Cariboo Island.



LEGEND

- SAMPLE LOCATION
- BATHYMETRY (10m)
- WATERCOURSE

COPPER CONCENTRATION mg/kg dry weight (<63 µm fraction)

- INFERRED CONCENTRATION <120 (< SedQCss)
- <120 (< SedQCss)
- 120 – 240 (<2x SedQCss)
- 240 – 600 (<5x SedQCss)
- 600 – 1,200 (<10x SedQCss)
- ≥1,200 (≥10x SedQCss)

0 1.5 3

1:70,000 KILOMETRES

REFERENCE(S)

1. WATERCOURSES OBTAINED FROM IHS ENERGY INC.
2. IMAGERY COPYRIGHT © 20170510 ESRI AND ITS LICENSORS. SOURCE: DIGITAL GLOBE. USED UNDER LICENSE, ALL RIGHTS RESERVED. IMAGE DATE: 20100411

DATUM: NAD83, PROJECTION: UTM10

CLIENT
MOUNT POLLEY MINING CORPORATION

PROJECT
MOUNT POLLEY MINE ECOLOGICAL RISK ASSESSMENT

TITLE
COPPER CONCENTRATIONS IN SURFACE SEDIMENT OF QUESNEL LAKE, 2014-2016

CONSULTANT

Golder Associates

YYYY-MM-DD	2017-06-06
DESIGNED	JVG
PREPARED	CD
REVIEWED	JVG
APPROVED	BGM

PROJECT NO.	CONTROL	REV.	FIGURE
1662612	22300	0	45

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IF THIS MEASUREMENT DOES NOT MATCH WHAT IS SHOWN, THE SHEET SIZE HAS BEEN MODIFIED FROM: ANSI A

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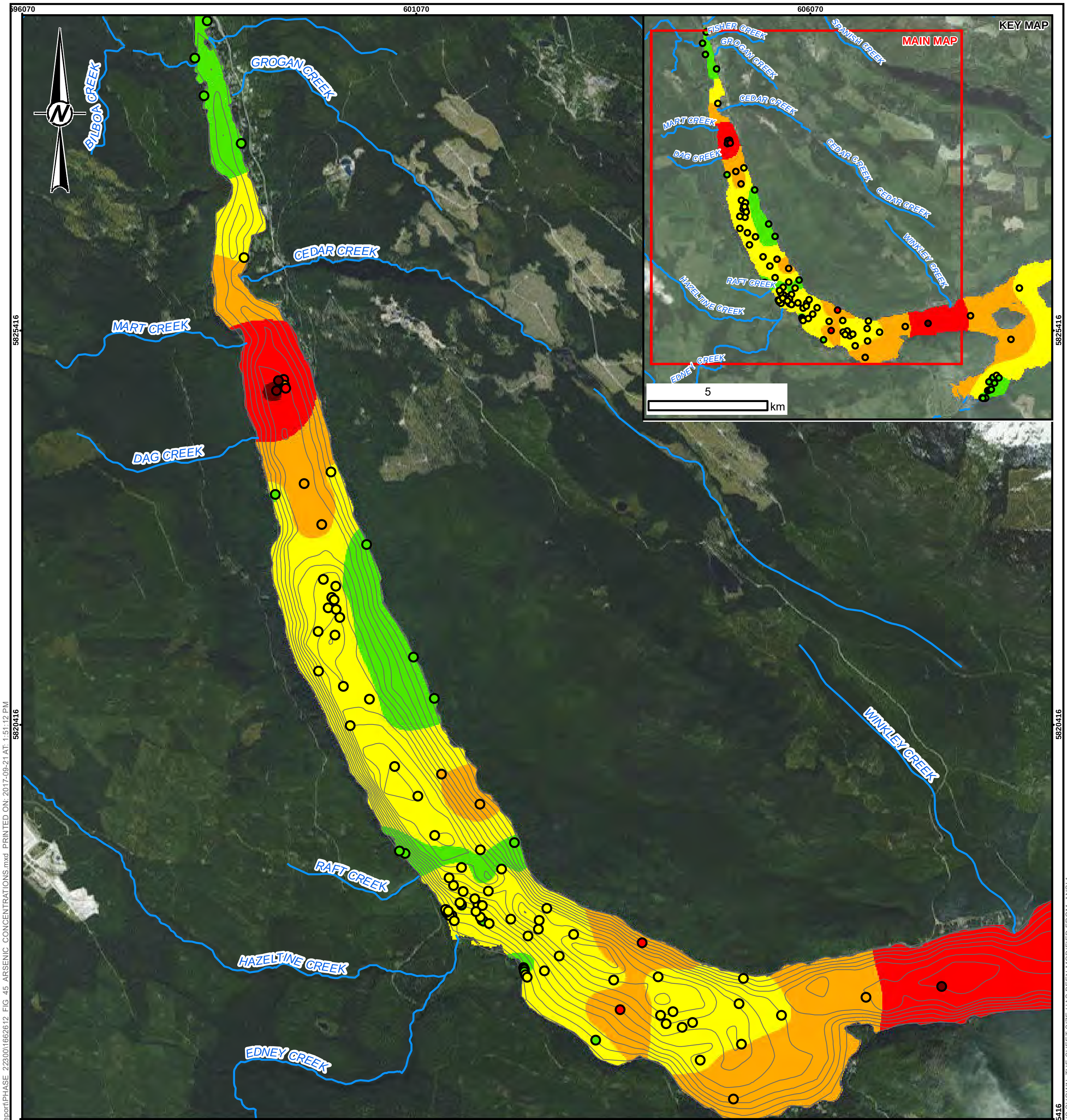

LEGEND

- SAMPLE LOCATION
- BATHYMETRY (10m)
- WATERCOURSE

ARSENIC CONCENTRATION mg/kg dry weight (<63 µm fraction)

- 0 – 11 (<SEDQCSS)
- 11 – 22 (<2X SEDQCSS)
- 22 – 55 (<5X SEDQCSS)
- 55 – 110 (<10X SEDQCSS)
- ≥110 (≥10X SEDQCSS)

0 1.5 3

1:70,000 KILOMETRES

REFERENCE(S)

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DATUM: NAD83, PROJECTION: UTM10

CLIENT
MOUNT POLLEY MINING CORPORATION

PROJECT
MOUNT POLLEY MINE ECOLOGICAL RISK ASSESSMENT

TITLE
ARSENIC CONCENTRATIONS IN SURFACE SEDIMENT OF QUESNEL LAKE, 2014-2016

CONSULTANT

Golder Associates

YYYY-MM-DD	2017-06-15
DESIGNED	JVG
PREPARED	CD
REVIEWED	JVG
APPROVED	BGM

PROJECT NO.	CONTROL	REV.	FIGURE
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4.6.1.5 Uncertainty Analysis

Data quality for sediment chemistry parameters was assessed by Minnow in Appendix A-7.2, and was considered to be acceptable for the use of interpretation and derivation of conclusions.

There are no historic (i.e., pre-breach) sediment chemistry data available for Quesnel Lake; as such, spatial reference comparisons were made to samples collected from Horsefly Bay, which was not considered to be impacted by the TSF embankment breach. It is possible that sediment near the mouth of the Hazeltine Channel and throughout the West Basin of Quesnel Lake exhibits natural variability and differences in sediment quality (e.g., local mineralization condition or particle size characteristics) relative to Horsefly Bay. Historical sediment data were available for Polley Lake and were used in combination with data from Bootjack Lake to define reference or background conditions. Thus the comparison for sediment in Polley Lake was likely to be representative of potential impacts from the TSF embankment breach.

Although sediment chemistry data were indicative of exposure conditions, concentration data do not provide information on potential effects beyond a cursory screening assessment. Chemistry data are compared to standards that provide a high degree of conservatism; there is therefore confidence that concentrations less than the $SedQC_{SS}$ are unlikely to present a risk. However, a substance may not present a risk simply because its concentrations exceed the criteria values; generic criteria and standards do not incorporate site-specific modifying factors, such as tolerances of resident species or bioavailability. There are natural background concentrations of metals that need to be considered. Overall, for chemistry measurements that reflect total concentrations rather than bioavailable fractions, the strength of association to the assessment endpoint is considered low.

As discussed in Section 4.6.1.2, differences in particle size distribution of sediment samples can have important implications on the interpretation of chemistry data. Laboratory analysis for metals in sediment were performed on the fine portion of the sediment sample only (<63 µm fraction). However, Minnow reported that 2014 sediment chemistry results were similar in the fine and coarser portions (<63 µm and <2 mm, respectively). Therefore, particle size distribution should not significantly affect the sediment chemistry results.

4.6.1.6 Comparison to Decision Criteria

Golder considered the patterns and gradients of sediment quality, and magnitude of exceedance of the CSR standards to evaluate the sediment chemistry line of evidence.

Copper—Copper was identified as the primary COPC because concentrations in water and sediment were above numerical standards following the breach.

- Although sediment concentrations throughout Polley Lake exceeded the $SedQC_{SS}$, measured concentrations were only 1.5-times reference concentrations. Bootjack Lake and pre-breach conditions in Polley Lake indicate that sediment concentrations are naturally elevated above the CSR criteria in these areas.
- Copper concentrations in the Hazeltine Channel sediment exceed the CSR standards and were higher than the reference concentration. Copper concentrations were highest in upper Hazeltine Channel in 2014, but much of the tailings-impacted materials has since been removed from the upper portion of the creek in conjunction with construction activities in 2015 and 2016. Concentrations in the sedimentation pond in lower Hazeltine remain less than 4-times the $SedQC_{SS}$ or less than 2-times the $SedQC_{TS}$.

- Approximately half the area in the West Basin of Quesnel Lake has copper concentrations below the $SedQC_{SS}$ and in the other half of the basin copper concentrations are less than 5-times the $SedQC_{SS}$. The most impacted areas are in the near-field and profundal zones, consistent with the pattern of debris flow and deposition of materials into the lake following the breach. Only a small portion of the lake bottom has copper concentrations greater than 5-times the $SedQC_{SS}$.

Arsenic—Arsenic was identified as a COPC only in sediment (i.e., not for water or tissue). Arsenic concentrations in Polley Lake and the Hazeltine Channel sediments marginally exceeded the $SedQC_{SS}$, but were within the range of reference conditions. This suggests that arsenic is naturally present in these areas at concentrations equivalent to the $SedQC_{SS}$. Arsenic concentrations in Quesnel Lake exceeded the $SedQC_{SS}$ by up to 1.7-times, but the spatial pattern did not reflect that observed for copper, which indicates that arsenic concentrations in Quesnel Lake sediment are not related to the breach, but rather to natural sources of variation. Therefore, although arsenic was identified as a COPC based on screening of the 2014-2015 data set, concentrations and spatial patterns do not support the identification of arsenic as a tailings-influenced parameter.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Sediment Chemistry	Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings	Compare sediment chemistry to CSR Schedule 9 standards (Column II, Sensitive)	Concentrations are not greater than sensitive standards	Polley Lake	Copper and arsenic concentrations exceed standards. Arsenic concentrations marginally exceed standards.
				Quesnel Lake	Copper and arsenic concentrations exceed standards.
				Hazeltine Channel (pre-restoration)	Copper concentrations exceed standards. Arsenic concentrations marginally exceed standards.
		Compare sediment chemistry data to the substance-specific background concentration	Concentrations are not greater than background.	Polley Lake	Copper concentrations marginally exceed background. Arsenic concentrations are within range of background.
				Quesnel Lake	Copper and arsenic concentrations exceed background.
				Hazeltine Channel (pre-restoration)	Copper concentrations exceeded background. Arsenic concentrations are within range of background.

4.6.2 Sediment Porewater Chemistry

Porewater chemistry in samples collected from Bootjack Lake (reference), Polley Lake, and Quesnel Lake was evaluated, with a focus on 2016 data. Porewater results were compared to reference concentrations and BC WQGs to assess whether porewater in deposited embankment breach materials is elevated in metals concentrations and/or potentially toxic to benthic invertebrates living near the sediment-water interface.

4.6.2.1 Study Design and Methodology

Minnow collected and processed sediment cores and grab samples from Polley Lake, Quesnel Lake, and Bootjack Lake, between 4 and 16 August 2016 (Appendix A-7.2, Appendix K-3). Most sediment grab samples were collected using a petite ponar, in accordance with technical guidance outlined in the British Columbia Field Sampling Manual (BCWLAP 2003) and Technical Guidance Manual for Environmental Effects Monitoring (EEM; Environment Canada 2012b). The top 3 cm of each grab was collected, homogenized, and shipped to ALS Environmental in plastic bags, where samples were centrifuged to separate porewater from sediments. The extracted porewater was analyzed for dissolved metals and particle size analysis was analyzed for the sediment.

Four sediment cores were collected (one from each of Bootjack Lake-reference and Quesnel Lake-exposed, and two from Polley Lake) for extraction and analysis of porewater. Cores were collected using a Tech Ops corer in accordance with technical guidance for gravity coring outlined in the British Columbia Field Sampling Manual (BCMWLAP 2003). Cores were extruded, sectioned, measured, photographed, and documented. Cores were sectioned into 1-cm increments that were centrifuged to separate the porewater and sediments. During processing, the cores were kept in a nitrogen gas-purged environment to prevent oxidation from atmospheric oxygen (Appendix K-3). Preservation of the samples was conducted in the field prior to shipping to ALS Environmental. Analysis for total metals was conducted on porewater samples from each 1-cm core interval. For additional information regarding core collection, refer to Appendix A-7.2 and Appendix K-3.

Porewater chemistry results were compared to BC WQGs. Although there are no porewater-specific BC WQGs, the freshwater 30-day average guideline for total copper (0.0015 – 0.01 mg/L, at hardness of 36.3 – 251 mg/L as $CaCO_3$) and freshwater interim guideline for arsenic (0.005 mg/L) were used as screening benchmarks. The details of the chemistry data were reported in Appendix A-7.2 and Appendix K-3.

4.6.2.2 Potentially Confounding Factors

For this line of evidence, mean porewater concentrations of arsenic and copper collected from areas exposed to deposited embankment breach materials were compared to reference areas and chronic BC WQGs; the two methodologies used for collection (ponar grab and sediment coring) convey different types of uncertainty, as summarized below.

- A relatively small number of ponar grab samples (*n*=3) and sediment core samples (*n*=1 in Bootjack Lake and Quesnel Lake and *n*=2 in Polley Lake) were collected. The small spatial representation of cores, and to some extent ponar grabs, does not allow for robust statistical analysis of the data, nor mapping of the exposure conditions across a gradient of representative site conditions.

- Samples collected by ponar represented only the top 3 cm of the substrate whereas the sediment core samples represented sediments from the surface of the substrate to approximately 20 cm below surface. Because benthic invertebrates may live deeper than 3 cm in sediment, sediment coring provides a more thorough depiction of potential exposure conditions, although the majority of taxa are expected to inhabit the near-surface sediment layer.
- The sediment core samples were processed (extracted, cut, centrifuged, and porewater was collected, preserved, and packaged) onsite and were kept in a nitrogen gas-purged environment, to prevent minimize oxidation from atmospheric oxygen. The ponar grab samples were homogenized onsite under normal atmospheric conditions, packaged, and shipped to ALS where they were centrifuged and the porewater was sampled. Oxidation of metals in the samples collected using a ponar sampler may have occurred, confounding the results of the analysis and making it difficult to directly compare samples.
- Samples collected by ponar were analyzed for dissolved metals analysis whereas samples collected by sediment core were analyzed for total metals. Although BC WQGs for copper are based on total copper concentrations, the dissolved fraction metals in water was considered a more realistic estimation of the potentially bioavailable portion of a metal in water. Because the sediment cores were analyzed for total metals, the estimated potential toxicity of the porewater was most likely overestimated. Also, because the differences in dissolved and total metal are unknown for the samples, a direct comparison of ponar sediment data and sediment core data was not possible.

4.6.2.3 Summary of Findings

Copper concentrations in sediment porewater exceeded chronic BC WQGs for the protection of freshwater aquatic life at all locations, including reference stations. Copper concentrations in sediment from the same locations also exceeded the $SedQC_{TS}$ (Table 2 and 4 of Appendix A-7.2). Table 41 summarizes porewater concentrations of these metals in each waterbody. The following sections discuss the results of porewater chemistry for the sediment COPCs (arsenic and copper).

Table 41: Sediment Porewater Concentrations of Copper and Arsenic in 2016 Sampling Program

Waterbody	Area - Method	Station	Copper WQG (mg/L)	Porewater Concentration (mg/L)		
				Hardness (as $CaCO_3$)	Copper	Arsenic
Polley Lake	Mid-depth—Ponar	BOL-1 (reference)	0.0018	44	**0.017**	0.0011
		POL-1	0.0072	180	**0.041** (83)	0.0022 (67)
		POL-2	0.008	201	**0.046** (92)	0.0032 (98)
	Deep—Ponar	BOL-2 (reference)	0.0015	36	**0.012**	0.0018
		POL-P1	0.009	225	**0.027** (77)	0.0028 (43)
		POL-P2	0.01	251	**0.010** (-18)	0.0015 (-18)
	Deep—Core	BOL-B2 (reference)	0.0026	65	**0.032**	0.0029
		POL-P2	0.0054	134	**0.059** (6)	0.0043 (37)
		POL-P2	0.0059	148	**0.076** (82)	**0.0058** (66)

Waterbody	Area - Method	Station	Copper WQG (mg/L)	Porewater Concentration (mg/L)		
				Hardness (as $CaCO_3$)	Copper	Arsenic
Quesnel Lake	Shallow—Ponar	LREF1	0.0032	80	**0.004**	0.0028
		LNF1	0.0040	99	**0.096** (184)	0.0017 (-42)
		LFF1	0.009	206	**0.012** (100)	**0.0073** (95)
	Deep—Ponar	PREF1 (reference)	0.0043	107	**0.012**	0.003
		PNF1	0.0054	135	**0.045** (116)	0.0029 (-3)
		PFF1	0.0062	155	**0.043** (113)	0.0019 (45)
	Deep—Core	QUL-NF	0.0045	112	**0.179**	**0.0052**

WQG = Water Quality Guideline.

Concentrations are the mean of multiple ponar grabs (n=3) or mean of 1-cm depth increments from a single sediment core sample (n=1). Concentrations for ponar and core samples are dissolved metals and total metals, respectively.

Values in brackets are the relative percent difference between the exposed and corresponding reference location.

Bold values indicate an exceedance of hardness-dependent BC WQG for copper or the interim BC WQG for arsenic (0.005 mg/L) for the protection of freshwater aquatic life.

Polley Lake

Porewater concentrations of copper and arsenic in Polley Lake sediments were higher than those observed at reference locations (Bootjack Lake), except for the POL-P2 (the deep exposed South Basin of Polley Lake) grab sample which was similar to reference. However, the magnitude of exceedance of reference conditions was small in all cases, with the maximum difference less than a factor of 3.

Porewater copper concentrations exceeded chronic BC WQG at all sampling locations in both Polley Lake and Bootjack Lake. This pattern is consistent with that observed for sediment (Section 4.6.1), in that Bootjack Lake and pre-breach conditions in Polley Lake indicate that sediment concentrations of copper are naturally elevated above the $SedQC_{SS}$ in these areas.

Porewater arsenic concentrations were consistently below the interim BC WQG, except in the POL-P2 core which marginally exceeded the WQG. As noted in Section 4.6.1, arsenic concentrations in Polley Lake sediments were within the range of reference conditions, suggests that arsenic is naturally present in these areas at concentrations equivalent to the $SedQC_{SS}$.

A subset of locations had core samples from both 2014 and 2016 that can be compared; the data also allow for evaluation of the vertical profile of contamination with sediment depth. Concentrations of copper in porewater from sediment cores collected from POL-P2 in 2016 were similar to the core collected in 2014 (Figure 7, Appendix K-3). This pattern was similar to that observed in the sediment chemistry profiles (Figure 6, Appendix K-3).

Concentrations of arsenic in porewater from sediment cores were higher and more variable with depth in 2016 compared to 2014 (Figure 7 of Appendix K-3). This pattern was relatively consistent with that observed in the sediment chemistry profiles (Figure 6 of Appendix K-3).

Quesnel Lake

Porewater concentrations of copper in sediments from exposed areas of Quesnel Lake (PNF1, PFF1, LNF1, LFF1) were higher than the Quesnel Lake reference locations (PREF1, LREF1) and exceeded BC WQG at all locations, including reference areas. Porewater concentrations of arsenic in the exposed areas of Quesnel Lake were lower than concentrations in reference areas and also lower than BC WQGs, with the exception of the shallow far-field sample (LFF1). This is consistent with that observed in the sediment chemistry line of evidence which suggests that spatial patterns of arsenic in sediment are not related to the TSF breach, but reflect natural sources of variation (Section 4.6.1).

Concentrations of copper in porewater from sediment cores collected in the near-field area were slightly higher and more variable with depth in 2016 compared to 2014 (Figure 11 of Appendix K-3). This pattern is in contrast with that observed in the sediment chemistry profiles, which showed lower concentrations that decreased with depth in 2016 compared to 2014 (Figure 10 of Appendix K-3). Copper porewater concentrations in 2014 exceeded chronic BC WQG.

Concentrations of arsenic in porewater from sediment cores were similar between 2014 and 2016 with little variation over core depth, consistent with that observed for sediment chemistry profiles (Figures 10 and 11, Appendix K-3).

4.6.2.4 Uncertainty Analysis

Data quality were assessed by Minnow in Appendix A-7.2 and K-3, and was considered to be acceptable for the use of interpretation. Each method of collection followed technical guidance outlined in the British Columbia Field Sampling Manual (BCWLAP 2003) and Technical Guidance Manual for Environmental Effects Monitoring (EEM; Environment Canada 2012b). However, differences in collection technique (ponar versus sediment core) and processing and preservations of samples (ponar samples sent to the lab for porewater extraction; sediment cores extracted and preserved onsite under inert atmospheric conditions) introduce variance in the data set. Difference in media (dissolved metals for ponar grab samples; total metals for sediment core samples) introduces additional variance. Although both analytical methods followed strict procedures, the difference in source media adds uncertainty when evaluating the data. Additional uncertainty comes from the limited spatial coverage and relatively small number of samples collected from each area (maximum number of samples collected from a sample location was n=3), as well as the lack of pre-breach data for comparison and natural differences in geochemistry between lakes.

Although sediment porewater chemistry data provides an indicator of potential exposure conditions, these data do not provide reliable information on potential effects. Directly applicable porewater guidelines are not available so chemistry data are compared to an interim BC WQG for arsenic and chronic BC WQG for copper. Both of these guidelines incorporate a high degree of conservatism, but also convey high uncertainty when applied to screening of porewater data, and do not consider site-specific factors, such as tolerances of resident species or bioavailability.

Given the above considerations, the strength of association to the assessment endpoint is considered to be low.

4.6.2.5 Comparison to Decision Criteria

The BC WQGs for copper and arsenic were included in this line of evidence to give context to the observed porewater concentrations. Golder notes that using an ambient water quality guideline for evaluating porewater provides a highly conservative screening tool. Porewater concentrations that are lower than the BC WQG indicate that exposure to porewater will not cause adverse effect to benthic organisms, but an exceedance of WQG in porewater samples does not necessary mean that adverse effects are present. Different organisms are exposed to porewater based on how they occupy and utilize the sediment habitat. A difference of more than 50% was considered to indicate an increased hazard to aquatic life which would be relevant for further evaluation using other lines of evidence

At all locations sampled, including reference locations, copper concentrations were greater than the chronic BC WQG for the protection for aquatic life. For all sample locations, copper concentrations were greater than reference locations by more than a factor of 1.5 except for the exposed South Basin of Polley Lake (POL-P2) sample, which exhibited lower concentrations of both copper and arsenic compared to the reference location in Bootjack Lake (BOL-B2).

The mean arsenic concentrations at all sampling stations in Polley Lake were below the interim BC WQG for the protection of freshwater aquatic life, except for one station (POL-P2). In Quesnel Lake, two stations marginally exceeded the interim BC WQG for arsenic, with the highest concentration observed at the exposed shallow far-field area (LFF1; mean arsenic concentration of 0.0073 mg/L). Porewater arsenic concentrations exceeded reference concentrations at some exposed stations, but not all. The infrequent and marginal exceedance of the BC WQG (which are highly conservative when applied to porewater) along with the low magnitude of elevation relative to reference conditions, indicate a very low hazard associated with porewater arsenic. Therefore, arsenic concentrations in porewater were not considered to reflect elevated exposure as a result of the TSF breach, consistent with that noted for sediment concentrations of arsenic.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Sediment porewater chemistry	Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Measure concentrations of metals in porewater from sediment samples.	The concentration in porewater is less than 50% elevated relative to reference or generally below ambient WQG for protection of aquatic life.	Polley Lake	The decision criterion was met for arsenic but not for copper.
				Quesnel Lake	The decision criterion was met for arsenic but not for copper.

4.6.3 Sequential Extraction

SRK (2015, Section 5.9) described the results of a sequential extraction test on sediments that was conducted in collaboration with Minnow. The purpose of these tests is to identify the degree to which different fractions of the total metals fraction are rendered accessible in progressively aggressive leaching conditions, as an indication of the relative bioavailability of the component fractions. Splits of sediment samples were also tested using a shake-flask method to determine the water soluble method (i.e., a different method to measure the same water soluble fraction described in Section 3.2.2.1). The sequential extraction method followed the general approach described by Tessier et al. (1979) with some minor modifications. The Tessier sequential extractions involve treating soil with a series of different solutions intended to leach out different fractions of the total metals concentrations in a sample. No single extract perfectly characterizes bioavailability, but the relative ease of extraction for different fractions provides a series of operationally-defined “fraction of bioavailability.” The method is an indirect evaluation of actual bioavailability, which to be definitive would require supplementation with bioaccumulation and/or direct toxicity testing (Peijenberg et al. 2007). In this particular application, the operationally-defined fractions were (in decreasing order of availability or dissociation potential):

- Adsorbed, but readily exchangeable (extracted with magnesium chloride)
- Associated with carbonates (extracted with sodium acetate)
- Associated with iron oxides (extracted with hydroxylamine hydrochloride in acetic acid)
- Associated with organic carbon (extracted with nitric acid and ammonium acetate)
- Residual (extracted with aqua regia)
- Figure 47 shows that the majority of copper in tailings that were deposited into the Hazeltine Channel or Quesnel Lake are associated the last two fractions that have relatively low potential to dissociate into porewater.

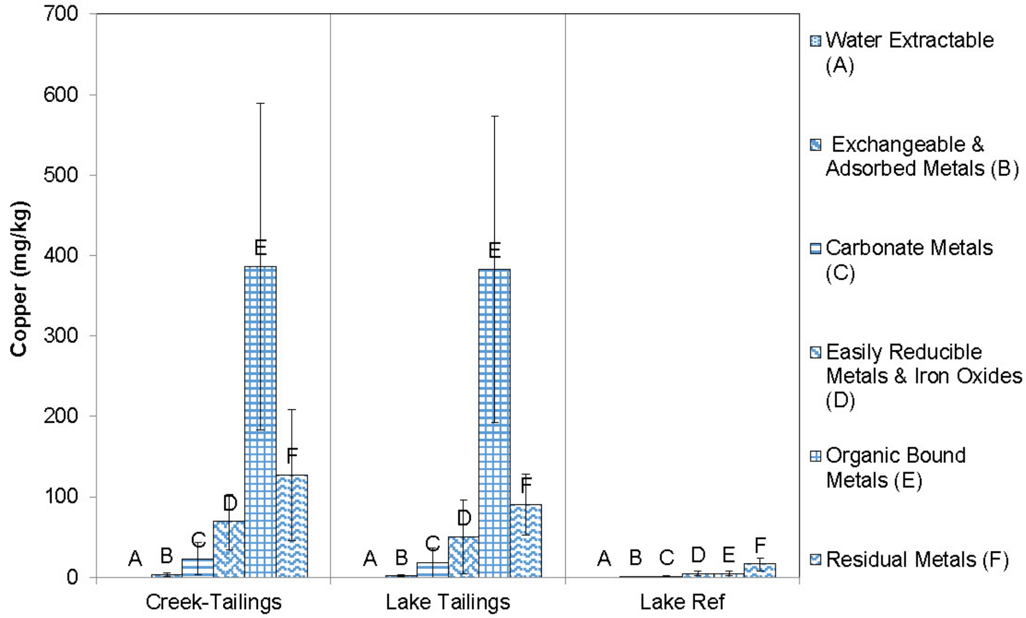

Figure 47: Summary of Sequential Extractions for Copper (from SRK 2015)

There are no formal decision criteria assigned for this line of evidence. Rather, the sequential extraction information provides supporting information with respect to the general bioavailability of copper and other elements in the tailings-influenced sediments. This information provides context to the observations of tissue bioaccumulation, sediment chemistry relative to guidelines, sediment toxicity, and other lines of evidence for which bioavailability is a toxicity modifying factor.

4.6.4 SEM-AVS

Sulphides act as a binding phase for many cationic metals in anoxic sediment, thereby reducing bioavailability of these contaminants. The simultaneously extracted metals- acid-volatile sulphides (SEM-AVS) method has been verified for five divalent metals (cadmium, copper, lead, nickel, and zinc), and one monovalent metal (silver; Wang and Chapman 1999). The ratio of SEM:AVS provides information about the likely bioavailability of metals in sediment samples. SEM:AVS of less than 1 indicate that sufficient sulphide exists to bind SEM, and metal toxicity is not expected (US EPA 2007). However, a ratio greater than 1 does not suggest that sediment toxicity will occur because AVS is only one of several binding sites and mechanisms that sorb cationic metals.

4.6.4.1 Study Design and Methodology

The sediment SEM-AVS sampling was conducted by Minnow concurrent with sample collection for other sediment and porewater chemistry (Sections 4.6.1 and 4.6.2). Sampling locations and sample collection methodology are summarized below. Details are provided in Appendix K-4[58].

- Samples for AVS and SEM analyses were not homogenized and were collected using procedures to limit sample oxidation[59]. Samples collected for TOC (and other concurrent analyses) were homogenized prior to being transferred to appropriate sample jars.
- The relevant analyses conducted included SEM (i.e., copper, lead, cadmium, zinc, and nickel[60]) and AVS according to standard procedures. Total organic carbon (TOC) was determined from the silt/clay fraction (<63 µm).
- The calculations for the sum of SEM, and SEM:AVS ratio, were calculated using the detection limit concentrations, where parameters were below the laboratory detection limit.

4.6.4.2 Summary of Findings

Figure 48 provides a summary of the minimum, maximum, and mean SEM:AVS ratios for each area compared to the decision criterion (SEM:AVS ratio = 1).

[58] See Appendix K-4 for rationale for sampling locations (Table 1), sample locations (Figure 1), and a summary of locations and methods (Table A.1).

[59] Note that a laboratory error resulted in some SEM-AVS samples being analyzed from sample aliquots that were collected for other concurrent analyses, which were not collected using these oxidation-minimizing procedures (Refer to Appendix C of Appendix K-4 for the laboratory corrective actions). The results of these analyses were not deemed to have affected the reliability of the results.

[60] Silver was not included in the standard SEM analysis and was not considered a parameter of interest.

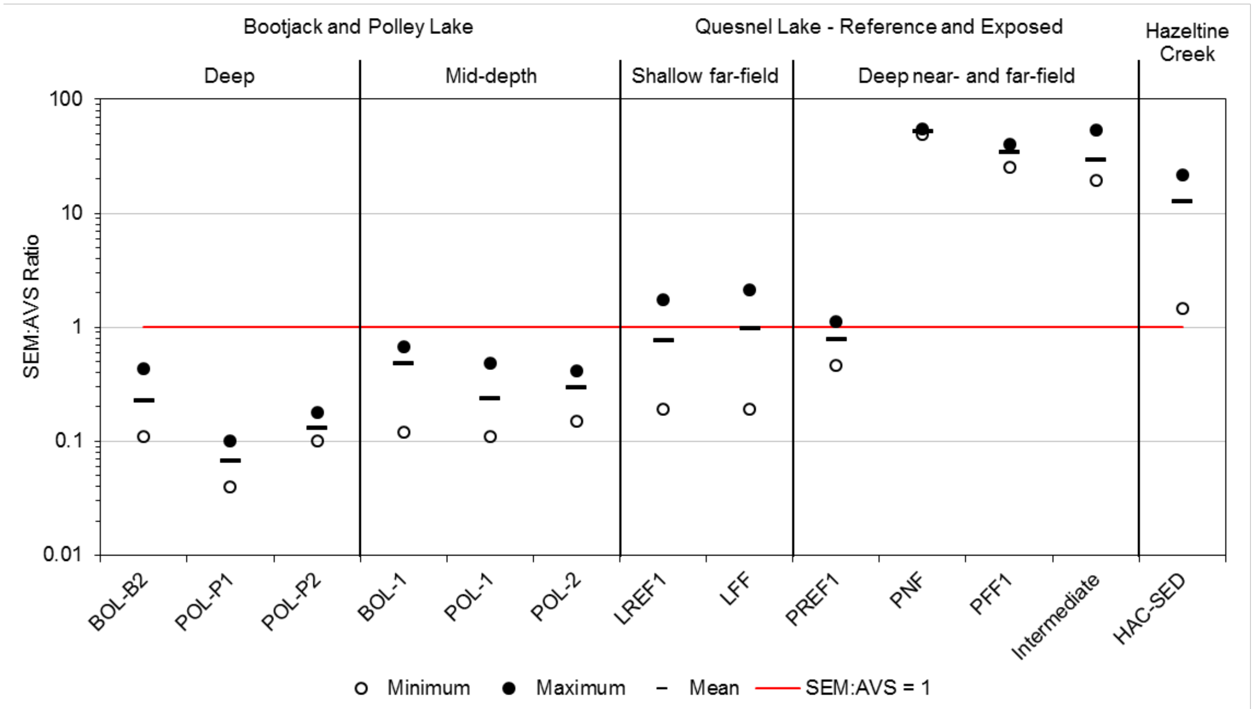

Figure 48: Summary of SEM:AVS Ratios for Sediments in Polley Lake, Quesnel Lake, and the Hazeltine Channel Sedimentation Pond

Note: *n*=10 intermediate locations with 1 sample each, *n*=5 replicates for HAC-SED, and *n*=3 replicates/area for all other areas. The decision criterion SEM:AVS ratio of 1 is shown by the red line. BOL = Bootjack Lake (reference); POL = Polley Lake; REF = Quesnel Lake reference; L = littoral; P = profundal; NF = near-field; FF = far-field.

4.6.4.3 Uncertainty Analysis

SEM-AVS analysis is a common approach supported by a body of literature with respect to its ability to predict an absence of toxicity related to copper. However, the method has lower utility when applied to oxidized environments (where sulphide concentrations are naturally low) (Di Toro et al. 1990). SEM concentrations greater than the available AVS concentrations indicate that AVS is likely insufficient to fully sequester the SEM, but does not necessitate that the excess SEM is bioavailable. In such a case, other ligands or mineral phases could be present and similarly bind SEM (e.g., organic carbon). Other lines of evidence are required to evaluate these conditions.

Sediments from deep areas of Quesnel Lake and from the Hazeltine Channel sedimentation pond had AVS concentrations that were low or below detection limits, therefore, application of the SEM-AVS method to these sediments may not be appropriate. Predictions of the potential for sediment toxicity cannot be made based on this method, however, previous geochemical investigations have indicated that the mobility of tailings-associated metals is expected to be low (SRK 2015).

Golder notes that there was an analytical error where samples that had not been appropriately collected and preserved. Details of the laboratory corrective action (a comparison of SEM-AVS results between non-preserved and preserved sample aliquots) are provided in Appendix K-4. Had the results been affected, a low-biased AVS result due to losses would be expected to over predict the potentially bioavailability of SEM.

4.6.4.4 Comparison to Decision Criteria

The decision criterion was the SEM:AVS ratio. A ratio below 1 indicates that the metals included in the calculation (which includes copper) are unlikely to be bioavailable to benthic organisms. The OC-normalized excess SEM concentration benchmark of 130 µmol/g was used as supporting information to evaluate whether sediment toxicity is expected. To evaluate results from the different sampling areas, the minimum, maximum, and mean SEM:AVS ratios were compiled. The monitoring locations were evaluated as their respective suite of replicate samples, whereas the 10 intermediate samples consisted of a single sample at each location. These intermediate locations were pooled to provide coverage of the deep areas from near- to far-field. The Hazeltine Channel sedimentation pond was included for completeness, but does not represent a natural aquatic environment.

In brief, the available SEM:AVS data showed:

- There was excess AVS in all samples collected from Polley Lake, and the ratios were consistent with those observed in the reference site (Bootjack Lake). This suggests that copper is unlikely to be contributing to sediment toxicity.
- There was excess AVS in most samples collected from the near-field littoral location in Quensel Lake, and the rations were consistent with its reference sample. This suggests that copper is unlikely to be contributing to sediment toxicity.
- There was marginally excess AVS in the profundal reference area in Quesnel Lake, but the ratio was closer to 1 than samples collected in Bootjack Lake. This is consistent with the fact that Quesnel Lake is much larger and therefore less likely to accumulate large amounts of organic material that will decay and contribute to sulphides. There was little to no AVS in samples collected from the exposed areas in Quesnel Lake, which means that copper that dissociates from the tailings will be more likely to be bioavailable because it is not being bound by AVS. This alteration in AVS-SEM would be expected to gradually return to the conditions in the reference area as organic carbon accumulates and sulphides begin to form.
- The sedimentation ponds in the Hazeltine Channel also have low AVS and therefore, an apparent excess of SEM metals. This is likely to due to the construction and maintenance of the ponds, which is sustaining an oxidized environment where sulphide generation would be naturally low.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
SEM-AVS	Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Determine the ratio of simultaneously extractable metals to acid volatile sulphide (SEM:AVS).	A SEM:AVS ratio of less than or equal to 1 indicates that the metals included in the calculation (which includes copper) are unlikely to be bioavailable to benthic organisms.	Polley Lake	Decision criterion met
				Quesnel Lake littoral (far-field)	Decision criterion met
				Quesnel Lake profundal	Decision criterion not met
				Hazeltine Channel (sedimentation pond)	Decision criterion not met

4.6.5 Sediment Physical Characteristics

Particle size distribution and TOC in sediment samples was evaluated as a line of evidence to assist in assessing the spatial extent of the influence of the TSF embankment breach. Particle size distribution data for tailings samples collected by SNC-Lavalin in Hazeltine Channel in 2014 indicated that tailings samples were mostly sand and silt. Observations made by Minnow after initial sediment sampling programs in 2014 indicated that sediments containing tailings were generally grey fines or orange and black sand, which was different than sediment dominated by brown or dark brown fines observed in reference areas (MPMC 2015).

Grain size also has the potential to influence the partitioning and distribution of sediment-associated contaminants. As discussed in Section 4.6.1, sediment chemistry data are based on <63 µm sediments regardless of bulk sediment grain size composition. Metal concentrations were shown to be similar between <63 µm and <2 mm sediment fractions analyzed in 2014 (MPMC 2015) and therefore, the technique for screening the sediment is considered to have little influence on sediment chemistry. Physical characteristics such as grain size and TOC can also influence habitat quality for benthic invertebrates, and these may need to be considered as confounding factors in the evaluation of sediment toxicity (Section 4.7.1) and benthic invertebrate community structure (Section 4.7.2).

4.6.5.1 Study Design and Methodology

As described in the sediment chemistry line of evidence (Section 4.6.1), all sediment samples collected for chemistry included measurement of grain size on bulk sediment and TOC on the <63 µm fraction. Refer to Section 4.6.1.1 for details on sampling methods and locations. The 2016 mean percentages of each grain size fraction for exposed areas (i.e., Polley Lake, Quesnel Lake littoral or profundal, near-field or far-field, etc.) were compared against the 2016 mean values for reference areas.

4.6.5.2 Summary of Findings

Profundal sediments in Quesnel Lake contained higher clay content in 2016 relative to 2014 and 2015, however this difference is inferred to be related to the relocation of some monitoring stations. Littoral sediments exhibited higher sand content in 2016 relative to 2014 and 2015, which is inferred to be related to sediment “washing” in the littoral zone (Appendix A-7.2) whereby flushing of overlying water liberates fine particles without disturbing the larger particle sizes[61]. Figure 49 presents the grain size composition and % TOC of sediments in 2016 in Polley Lake, Quesnel Lake littoral and profundal, and the Hazeltine Channel and corresponding reference areas and spatial comparisons are discussed below.

[61] Heavier, coarser sediments deposit first; finer, lighter particles travel farther.

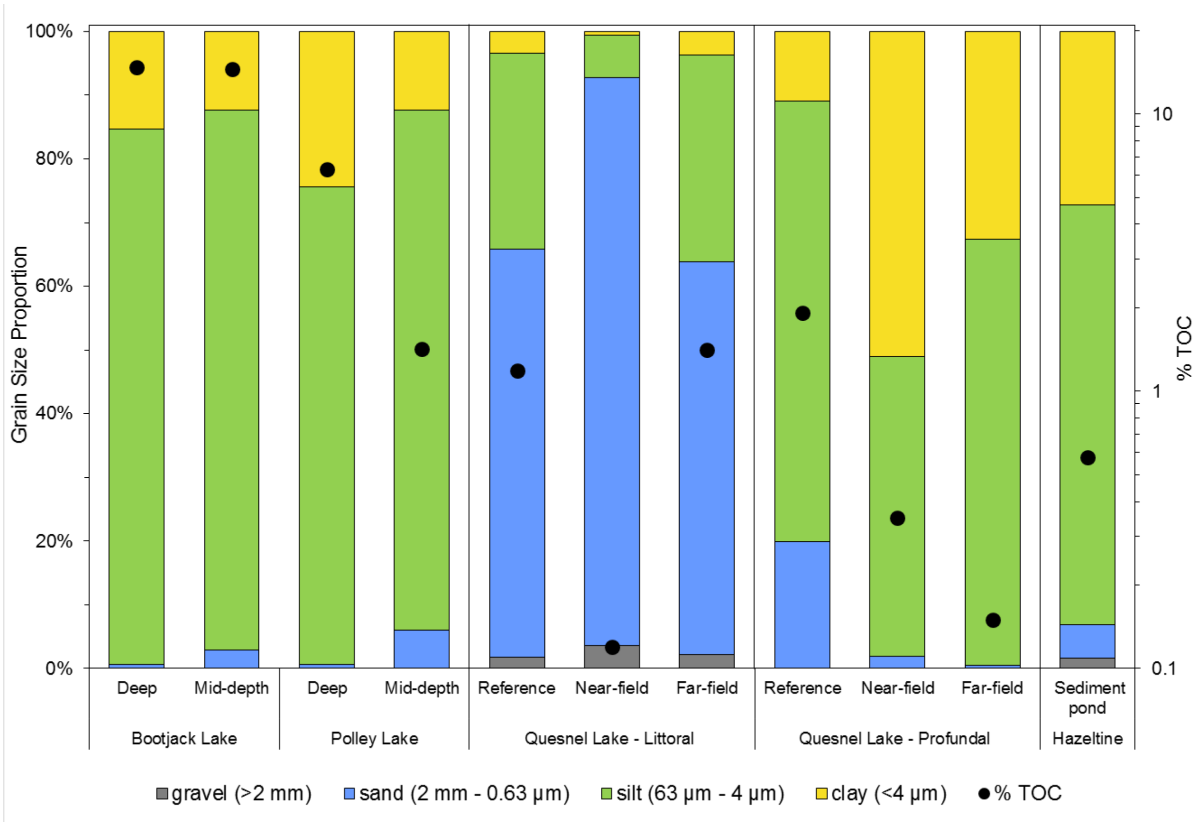

Figure 49: Sediment Grain Size Distribution and TOC from Exposed and Reference Areas, 2016

Note: Data are mean values from samples collected in 2016. For Polley Lake, data are average of north and south areas. No reference area for Hazeltine. Note logarithmic axis for % TOC.

Polley Lake

Sediment samples collected in 2016 consisted almost entirely of silt and clay (>94%), which was similar to samples collected in 2014 and 2015 and to reference sediment samples collected in Bootjack Lake in 2016 (>97% silt and clay; Figure 49). Sediment grain size composition was similar between both north and south areas, and deep and mid-depth areas sampled in Polley Lake.

In 2016 samples, the TOC content of mid-depth sediments from Polley Lake was similar in the north and south areas (1.4%) and was lower than in deep area sediments (5.1-7.6%). Bootjack Lake reference sediments have much higher TOC (15%) compared to current conditions in Polley Lake, but historically Polley Lake had higher TOC (~17-21%; Table D.1 and D.2 of Appendix A-7.2).

Hazeltine Channel

Sediment samples collected from the sedimentation pond in lower Hazeltine Channel in 2016 were similar to those in 2015 and consisted primarily of silt and clay (93%). The Hazeltine Channel pre-breach sediment (samples collected between 1995 and 2013) was primarily sand and silt (Table D.16 of Appendix A-7.2) with a lower proportion of fines compared to 2016 results for the sedimentation pond. The TOC content in sediments from the Hazeltine Channel sedimentation pond was very low (<1% in 2014–2016) in general and in comparison to historical data (9–13%).

Quesnel Lake – Littoral

Sediment collected from the littoral near-field area of Quesnel Lake in 2016 was predominantly sand (89%) and was similar to 2015 samples (82% sand). However, these samples had a higher sand content than those collected from this area in 2014 (49%; Appendix A-7.2), as well as the reference sediment from Horsefly Bay (64% sand). This difference in sediment composition may be a result of washing of fine sediment from these locations between 2014 and 2016. The particle size distribution of sediment collected in the littoral far-field area was similar between years (62% sand) and was similar to the reference area in Horsefly Bay.

The TOC content for littoral, near-field sediments was very low (0.12%) in general and compared to reference sediment (1.2%). In far-field sediments TOC was higher (1.4%) and more similar to the reference area (Figure 49).

Quesnel Lake – Profundal

Sediment collected from the profundal near-field and far-field areas of Quesnel Lake in 2016 were predominantly silt and clay (98%), with a slightly finer-grained composition than the reference area (80% silt and clay). The clay content in the 2016 samples (32–51%) was higher than in 2014 and 2015 (15%–36%), possibly due to the relocation of two sediment stations in 2016 (PNF-4 and PNF-5; Appendix A-7.2). The TOC content of near-field and far-field sediments was very low (0.15–0.35%), including in comparison to the reference area (1.9%).

4.6.5.3 Comparison to Decision Criteria

The decision criterion for the sediment physical characteristics line of evidence was a qualitative comparison of conditions between areas.

Overall, there were no large differences in the grain size distributions throughout Polley Lake (and Bootjack Lake) sediments that would warrant consideration of this characteristic as a potential confounding factor for these water bodies. The TOC content in Polley Lake sediment was lower in mid-depth areas relative to deep areas and was lower than in Bootjack Lake. These differences in TOC may have the potential to influence interpretation of biological results; however, these concentrations are considered to be within a normal range for natural sediments.

In littoral areas of Quesnel Lake, near-field sediments were of coarser composition and had a very low TOC content (<1%) compared to far-field and reference areas, both of which may be confounding factors for the interpretation of biological effects. Far-field and reference samples from littoral areas had similar grain size distribution and TOC content. The profundal areas of Quesnel Lake had a marginally finer-grained composition in near-field and far-field areas relative to the reference area, but were not sufficiently dissimilar to indicate that grain size may be a confounding factor. However, TOC content in near-field and far-field profundal areas was very low (<1%) compared to the reference area was identified as a potential confounding factor.

The grain size distribution and TOC content of sediment from the Hazeltine Channel has changed substantially from conditions prior to the breach, as a direct result of the scouring of material along the channel. However, due to the ongoing remediation in the creek, there has been only limited evaluation of the potential for biological effects (i.e., sediment toxicity testing in 2014, but no benthic community monitoring).

4.6.5.4 Uncertainty Analysis

Sediment samples were collected with standardized field protocols and physical characteristics were measured with standard analytical methods with robust QA/QC practices. Data were compared to background conditions, which provides a high degree of certainty that if physical conditions are substantially different from background data then these physical conditions can potentially be confounding factors in the interpretation of biological effects. Samples were collected with a sufficient density provide confidence that the data were representative of site conditions.

There are no numerical standards or guidelines for physical parameters. Interpretation focuses on comparison between exposed and background areas, supported by literature that provides context about the likely tolerance of major taxa to low organic carbon and predominantly fine-grained samples (discussed in Section 4.7.1).

Sediment substrate can influence habitat quality for benthic invertebrates, but this information alone does not provide a strong relationship to the assessment endpoint. Therefore, the strength of association to the assessment endpoint is considered weak (low).

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Sediment Physical Characteristics	Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings	Compare particle distribution and organic carbon content between impacted area and background conditions	Grain size distribution and organic carbon are not substantially different (e.g., >20%) from background	Polley Lake	Grain size distribution is consistent with background. Organic carbon content is reduced in Polley Lake “deep” areas, but not in “shallow” areas.
				Quesnel Lake	Grain size distribution is substantially different and organic carbon content has been reduced in three of the four areas. Quesnel Lake littoral far-field has similar grain size and organic carbon content as background.

4.7 Effects-Based LOE: Benthic Invertebrates

4.7.1 Sediment Toxicity Testing

Laboratory-based toxicity tests are commonly used tools that provide a direct measure of the potential for biological effects in individual organisms, under the assumption that laboratory exposures mimic site conditions. Toxicity tests indicate whether the mixtures of contaminants found at a site elicit responses to sensitive freshwater organisms under controlled laboratory conditions. These tests account for many of the site-specific factors (bioavailability, speciation, substrate conditions) that mediate the toxicity of contaminants in field sediments. However, there is the potential for confounding factors to influence the interpretation of results in these tests; the specific causes of biological variability in organism responses often remains the largest source of uncertainty in controlled studies. Not only does variability exist across replicates within a test, but also among tests, and it can influence the ability to detect meaningful differences in biological responses between tested sediments.

4.7.1.1 Study Design and Methodology

A sediment toxicity testing program was initiated in 2014 following the TSF embankment breach to evaluate the bioavailability and toxicity of COPCs, as well as to document changes to physical conditions of sediment that may affect the capacity of sediment to support sediment-dwelling organisms. Between August and October 2014, surface sediment samples were collected by Minnow from exposed areas in Polley Lake (mid-depth and deep), along the Hazeltine Channel (upper, mid, lower, at mouth), within the West Basin of Quesnel Lake (littoral and profundal areas both near-field and farther afield), and from reference areas in Bootjack Lake and Quesnel Lake (Horsefly Bay and the North Arm)[62]. All samples were tested with two sensitive invertebrate test species using the following standard toxicity tests:

- 10-d survival and growth of the midge *Chironomus dilutus* (Environment Canada 1997)
- 14-d survival and growth of the amphipod *Hyalella azteca* (Environment Canada 2013a)

Both *C. dilutus* and *H. azteca* are commonly used test species with standardized tests protocols and have ecological relevance to the site, particularly for depositional environments such as lakes. *Hyalella azteca* is a surrogate for resident crustaceans and other epifauna that live at the sediment-water interface, whereas *C. dilutus* is a surrogate for chironomids and other insect taxa that typically burrow into sediment.

Organism responses in exposed sediments were compared to both the concurrent laboratory negative control and reference samples[63] because of the potential for non-chemical factors to influence organism response in chronic tests. Responses in reference and exposed sediments were normalized to control responses to facilitate comparisons between samples run in different test batches. Toxicity was identified as environmentally meaningful in cases where survival or growth were reduced by more than 20% relative to laboratory control and reference sediment. In cases where toxicity was identified, results were also evaluated relative to the spatial distribution of sampling areas (and linkage to tailings influence).

In 2015, testing of Quesnel Lake sediment was conducted to confirm the general patterns of toxicity observed in 2014 and additional experimental treatments were included to elucidate the respective roles of physical and chemical factors[64]. The focus of testing was on sediment from the near-field profundal zone, near the mouth of the Hazeltine Channel, as sediments from this area exhibited the greatest frequency of toxicity responses in the 2014 tests, exhibited elevated copper concentrations, and exhibited altered substrate composition relative to reference samples (MPMC 2015). The design of this study was focussed on elucidating potential causes of responses rather than to provide a broad spatial profiling of responses.

- The standard protocols for *C. dilutus* and *H. azteca* were retained and included the addition of a clay control sediment to mimic the uniformly fine materials deposited in Quesnel Lake.

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[62] See Figures 4.1 through 4.5 of MPMC 2015 Appendix E for sediment sampling locations

[63] There were two deep reference samples from Bootjack Lake and two littoral and profundal reference samples (Horsefly Bay-Ref1 and North Arm-Ref2) from Quesnel Lake. There was no reference sample for the Hazeltine Channel.

[64] Details of 2015 sampling and test methods are provided in Appendix L-4.

- The design also incorporated additional treatments in which each test sediment (i.e., laboratory and clay control, near-field exposed, and reference) was supplemented with 2% organic carbon to evaluate whether endpoint responses would be influenced by addition of additional food sources.
- Another treatment comprised of equal proportions (50:50) of near-field and reference sediment was tested to address uncertainty related to how sediment toxicity may be influenced by the incorporation of natural sediment.
- In 2016, toxicity tests were repeated on sediment from Polley Lake, Bootjack Lake, and Quesnel Lake (littoral and profundal)[65]. The laboratory control sediment was supplemented with 5% organic carbon to minimize the disparity in TOC relative to reference sediments, and this better represented field reference conditions.
- Using the standard protocols, the following experimental treatments and additional test sediments were included to further examine the influence of physical factors and physical/chemical gradients:
  - clay control sediment that was pH-stabilized
  - Quesnel Lake profundal reference sediment with TOC removed[66] (PREF1-TOC)
  - 10 “intermediate” samples representing a spatial gradient in the profundal area of the West Basin of Quesnel Lake
  - a laboratory-prepared gradient (or dilution series) consisting of near-field and reference sediment from the profundal area of Quesnel Lake, ranging from 10 to 90% proportions of each type
- A subset of samples (Polley Lake deep, Quesnel Lake profundal, near-field) were evaluated in a 28-d survival and growth test for *H. azteca* (US EPA 2000; Ivey et al. 2016) at the request of stakeholders. The 28-d tests were run concurrently with the 14-d tests. This longer-term term is an extension of the standard 14-d protocol and provided data with respect to addressing uncertainties related to endpoint sensitivity for the different test durations.

The Hazeltine Channel sediment was not included in the toxicity tests conducted in 2015 and 2016 as the creek was under reconstruction and was not considered to be fish habitat.

4.7.1.2 Summary of Findings

Figure 50 and Figure 51 provide an overview of available sediment toxicity data for the different waterbodies. A brief explanation of how the figures were developed is appropriate:

- All results were normalized to the performance of their respective negative controls. This normalization reduces the influence of inter-batch variability in organism performance on the overall patterns in toxicity data and effectively converts the toxicological endpoints which have different units (e.g., percent survival, mg/day growth) to a numerical scale.

[65] Details of 2016 sampling and test methods are provided in Appendix L-5.

[66] Sediment was heated to 455°C to remove TOC using a loss on ignition method

- A value of 1.0 on this scale means that the individual sample had a toxicological performance equal to the negative control. A value of 1.2 means that the sample had a toxicological performance that was 20% greater than the negative control, while a value of 0.8 means the sample had a toxicological performance that was 20% less than the negative control.
- All available data are presented on Figure 50 and Figure 51 on the same scale, irrespective of whether the sample originated from an impacted or reference area, or was part of the experimental sample manipulations to investigate the relative influence of low organic carbon.
- Data represent the mean performance of an individual sample. For 2014 samples, this was the mean of five laboratory replicates on a composite field sample. For 2015 samples, this was the mean of five field replicates. For experimental samples, only one replicate was tested.
- Decision criteria are highlighted as lines. The black dotted lines represents a value of 0.8 (i.e., a 20% reduction relative to the negative control. The red line represents a 20% reduction relative to the average of all reference data from a given area for a given year.

A summary of the key findings for each waterbody is summarized below.

Polley Lake

In 2014 and 2016, Polley Lake sediments indicated no instances of significantly reduced survival or growth (i.e., ≤20%) of either *C. dilutus* or *H. azteca* relative to concurrent laboratory controls and reference samples from Bootjack Lake, with the following exceptions:

- There was a modest reduction in *C. dilutus* survival (22-29%) in the south, mid-depth sample (POL-2) in 2014
- There was a modest reduction in *H. azteca* growth (28-38%) in the south, mid-depth sample (POL-2) in 2014 and both mid-depth samples (POL-1, -2) in 2016

Polley Lake sediment was not tested in 2015. Overall, the toxicity data indicate low effects in Polley Lake sediment.

Hazeltine Channel

Toxicity for both *C. dilutus* and *H. azteca* was greatest in the Hazeltine Channel samples, whereas toxicity responses of a smaller magnitude were observed in Polley and Quesnel Lake samples. Survival of *C. dilutus* was significantly reduced in samples from both upper and lower reaches of the Hazeltine Channel (by ~30–60%), but was not reduced in the sample at the mouth of the Hazeltine Channel. Growth of *C. dilutus* was not adversely affected in samples with reduced survival, but was significantly reduced in the sample at the mouth (by 35%). Survival and growth of *H. azteca* was significantly reduced in two of four samples from the Hazeltine Channel (i.e., upper and at the mouth). The strongest toxicity response in both test species were observed for the upper Hazeltine sample, which exhibited a >50% reduction in multiple endpoints. This sample exhibited coarser substrate (70% sand) and elevated copper relative to other creek samples, and the sample contained no detectable organic carbon (<0.1%).

Quesnel Lake

Results of the 2014 testing for Quesnel Lake indicated no adverse effects to either species for near-field, far-field, and far-far-field samples from the littoral zone relative to control and reference samples, with the exception of significantly reduced survival and growth of *C. dilutus* in one near-field sample. No consistent spatial patterns in the growth of *C. dilutus* were evident. For the profundal zone, survival and growth of *C. dilutus* were significantly reduced in the near-field sample and survival was reduced in one far-field sample. No effects to survival of *H. azteca* were observed, other than a modest reduction (22%) in the near-field sample relative to the control. Growth of *H. azteca* was significantly reduced relative to both reference samples only in one far-field sample. The apparent reduction in growth for other samples was only relative to one reference sample for which a high growth rate was measured relative to the second reference sample. Growth of *H. azteca* in littoral and profundal samples tended to increase with distance from the near-field area, likely in association with increasing TOC (MPMC 2015).

The 2015 investigation of toxicity of near-field sediment from the profundal area of Quesnel Lake indicated no effect to the survival endpoint for either *C. dilutus* or *H. azteca*, which differed from the 2014 results. However, the significant reduction in growth previously observed for both test organisms was confirmed in 2015, with growth being the more sensitive test endpoint in the impacted sediment tested in 2015. Significant effects on growth in the near-field sediment were eliminated with the addition of TOC or mixing the samples (50:50) with reference sediment.

The 2016 testing showed no adverse effects to *C. dilutus* for near-field and far-field samples from the littoral and profundal zones relative to control and reference samples, with the exception of significantly reduced growth in the near-field profundal sample, consistent with previous results. Growth effects on *C. dilutus* were reduced with increasing dilution of the near-field profundal sample (i.e., increasing proportion of reference sediment). Significant growth effects were observed until the sample was diluted to 20% with reference sediment. Survival of *C. dilutus* was reduced (32 and 43%) relative to reference in two of the 10 “intermediate” samples that covered a spatial gradient in the profundal zone and growth was reduced (37-69%) in all but two samples.

In 2016, survival and growth of *H. azteca* were significantly reduced in the near-field samples (littoral and profundal) and the far-field profundal sample, but were not affected in the far-field littoral sample. Survival of *H. azteca* in near-field profundal sediment improved to 80% (equivalent to control acceptability criterion) when the sample was diluted to 60% sample (v/v). Growth remained significantly reduced until the near-field sample was diluted to 20% sample (v/v). Survival of *H. azteca* was reduced (29 to 90%) relative to reference in seven of the “intermediate” samples and growth was reduced (54 to 84%) in nine samples.

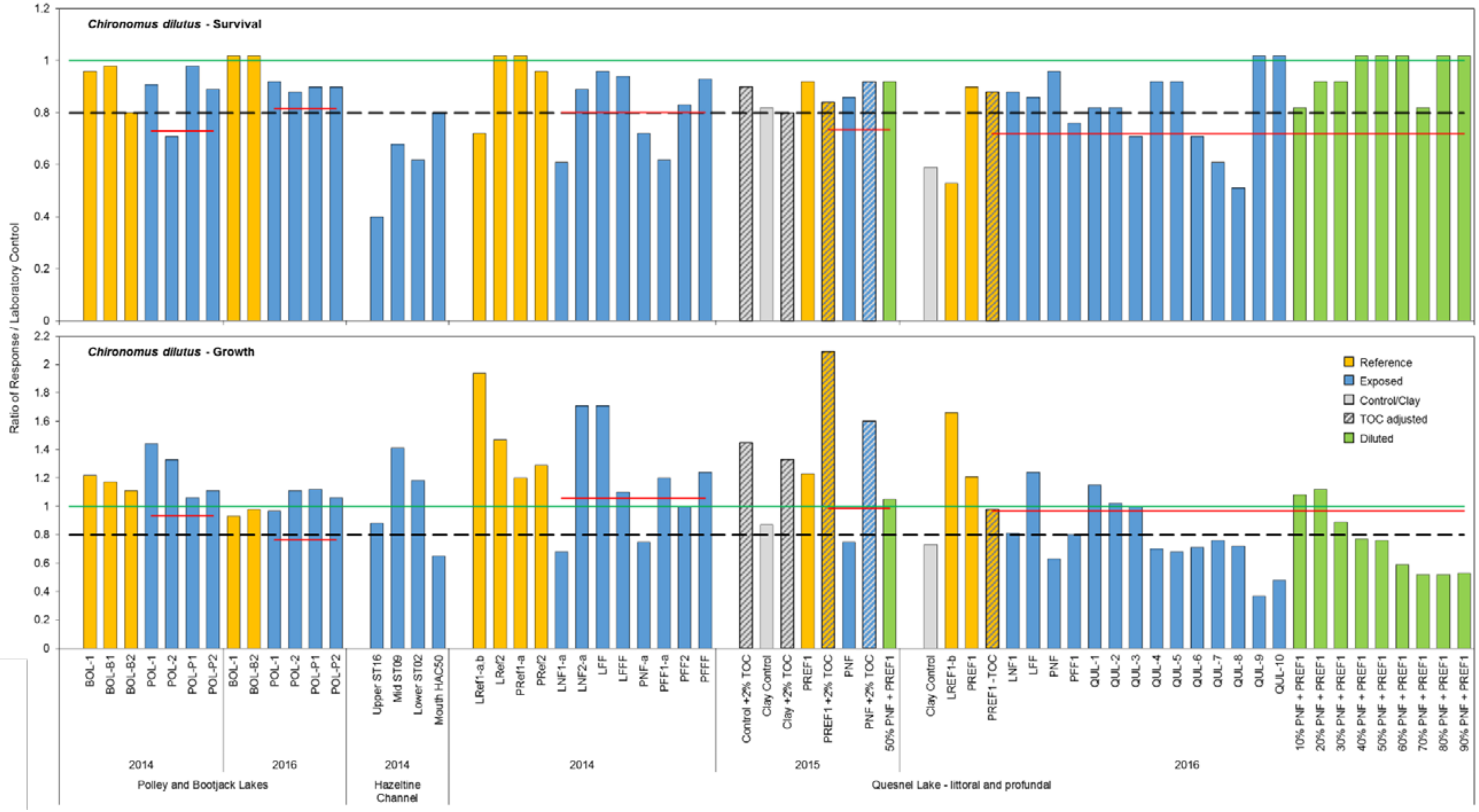

Figure 50: Survival and Growth (as mg dry weight) of Chironomus dilutus in Reference and Exposed Sediments Relative to Laboratory Control

Note: Results based on mean response normalized to concurrent laboratory control. 2014 based on 5 laboratory replicates per station, "-a" = area mean from 5 stations; 2015-2016 based on 5 field replicates, but 1 replicate for gradient and 2016 diluted samples. , "-b" = excluded from reference mean due to low survival, which enhanced growth of survivors. BOL = Bootjack Lake (reference); POL = Polley Lake; REF = Quesnel Lake reference (Horsefly Bay-Ref1, North Arm-Ref2); L = littoral; P = profundal; NF = near-field; FF = far-field; FFF = far-far-field; TOC = total organic carbon; QUL-1 to -10 = "intermediate" spatial gradient.

Solid green line—ratio of 1.0 is equivalent response to laboratory control; ratio >1.0 indicates higher survival or growth than control

Dashed black line—ratio of 0.8 is equivalent to 20% effect relative to laboratory control; ratio <0.8 indicates >20% effect relative to control.

Shorter red lines—ratio less than value indicates >20% effect relative to mean response of all reference samples for that year and lake, except where noted.

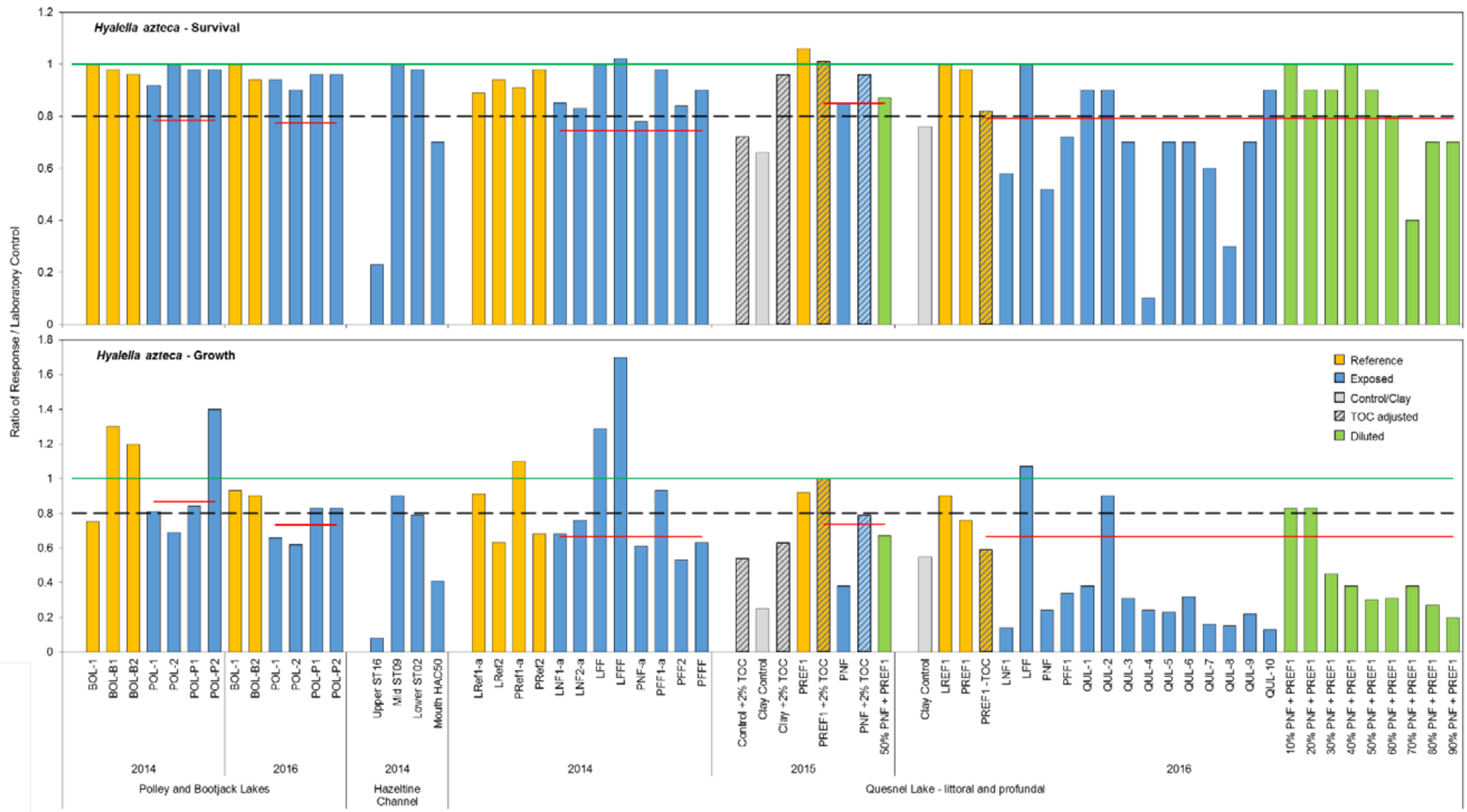

Figure 51: Survival and Growth (as mg dry weight) of Hyalella azteca in Reference and Exposed Sediments Relative to Laboratory Control

Note: Results based on mean response normalized to concurrent laboratory control. 2014 based on 5 laboratory replicates per station, "-a" = area mean from 5 stations; 2015-2016 based on 5 field replicates, but 1 replicate for gradient and 2016 diluted samples. BOL = Bootjack Lake (reference); POL = Polley Lake; REF = Quesnel Lake reference (Horsefly Bay-Ref1, North Arm-Ref2); L = littoral; P = profundal; NF = near-field; FF = far-field; FFF = far-far-field; TOC = total organic carbon; QUL-1 to -10 = "intermediate" spatial gradient.

Solid green line—ratio of 1.0 is equivalent response to laboratory control; ratio >1.0 indicates higher survival or growth than control

Dashed black line—ratio of 0.8 is equivalent to 20% effect relative to laboratory control; ratio <0.8 indicates >20% effect relative to control.

Shorter red lines—ratio less than value indicates >20% effect relative to mean response of all reference samples for that year and lake.

Evaluation of Longer-term Toxicity Tests

A longer term, 28-d test with *H. azteca* was conducted with a subset of exposed and reference samples concurrent with the standard 14-d tests. Results are presented in Table 42 and summarized below.

- Growth of *H. azteca* in the Polley Lake south, deep sample (P2) showed a slightly increased effect size in the 28-d test (31% reduction) compared with the 14-d test (≤20% reduction), relative to the corresponding reference responses. Survival was not adversely affected in this sample for either test duration.
- Survival of *H. azteca* in the Quesnel Lake near-field profundal sample was similar in both tests and was significantly reduced in the 14-d (47% reduction) and 28-d (50% reduction) tests, relative to the corresponding reference responses. Growth of *H. azteca* showed a slightly increased effect size in the 14-d test (68% reduction) compared with the 28-d test (59% reduction), relative to reference responses.

Table 42: Comparison of 14- and 28-d Survival and Growth Tests with *Hyalella azteca*, 2016

Waterbody – Sample		14-d Test				28-d Test			
		Survival (%)		Growth (mg dw)		Survival (%)		Growth (mg dw)	
		mean	CV	mean	CV	mean	CV	mean	CV
Control Sediment		100 ± 0	0%	0.29 ± 0.02	7%	88 ± 9.6	11%	0.73 ± 0.06	8%
Bootjack Lake	Reference BOL-B2	94 ± 8.9	9%	0.26 ± 0.02	8%	98 ± 5	5%	0.8 ± 0.07	9%
Polley Lake	POL-P2	96 ± 5.5	6%	0.24 ± 0.04	17%	100 ± 0	0%	0.55 ± 0.04	7%
Quesnel Lake profundal	Reference PREF1	98 ± 4.5	5%	0.22 ± 0.02	9%	95 ± 10	11%	0.41 ± 0.16	39%
	PREF1-TOC	82 ± 16	20%	0.17 ± 0.02	12%	78 ± 17	22%	0.47 ± 0.09	19%
	Near-field	52 ± 16	32%	0.07 ± 0.02	29%	48 ± 17	36%	0.17 ± 0.01	6%

Data are mean ± standard deviation based on 5 field replicates for 14-d test and 4 field replicates for 28-d test. Growth is measured as average individual dry weight. CV = coefficient of variation.

The sensitivity of the survival endpoint was similar between the 14- and 28-d tests. This is consistent with observations by Norberg-King et al. (2006) who showed that the minimum detectable difference and coefficient of variation (CV) for survival in *H. azteca* did not vary with length of test (28- versus 42-d). It is noted that growth is typically a more sensitive endpoint than survival, but often more variable (US EPA 2000; Norberg-King et al. 2006). Consideration of endpoint variability as it relates to the ability to detect meaningful differences between samples is discussed in detail in a subsequent section. Evaluation of endpoint sensitivity between the 14- and 28-d tests should also consider the variability in endpoints by examining the CV, which is based on average responses across test replicates. The CVs for survival and growth responses in *H. azteca* were low (<20%) in the control, Bootjack Lake, and Polley Lake sediments for both test durations. In contrast, growth in the Quesnel Lake reference sample was more variable in the 28-d test (39%) than the 14-d test (CV 9%), while the opposite was observed for the near-field, exposed sample. This variability in growth endpoints in both reference and exposed sediments increases the uncertainty in the magnitude of effect observed for both test durations. Overall, inclusion of a longer test with *H. azteca* did not show a meaningfully increased sensitivity in response relative to the standard test duration.

4.7.1.3 Comparison to Decision Criteria

The decision criteria for the sediment toxicity line of evidence were selected based on toxicological endpoints for lethal and sub-lethal effects measured in the toxicity tests. Specifically, the selected criterion is that toxicological performance is not reduced by more than 20% relative to reference samples (or control if no appropriate reference sample was available). Comparisons were made preferentially to reference sediment for specific areas, where such can be well-matched in terms of substrate type or location of sampling. The acceptable effect size for discriminating negligible to low responses from moderate to high responses was set to 20% adverse response. This effect size is consistent with BC CSR policy concerning the interpretation of aquatic toxicity test data in risk assessments, and is also consistent with emerging technical guidance for Contaminated Sites Approved Professionals. The approach is also consistent with BC MoE's preference for use of specified effects levels (e.g., IC_x/EC_x values) for estimating risks to ecological receptors relative to the use of statistical significance measures (BC MoE 2012b). Additional support for the prioritization of EC_{20}/IC_{20} results comes from the following:

- The US EPA has selected EC_{20} values to be used to estimate a low level of effect that would be statistically different from control effects, yet not so severe as to be expected to cause chronic impacts at the population level (US EPA 2013).
- Ecological risk assessment guidance recommends the use of EC_{20}/IC_{20} results as a permissible level of effect. For example, an effects level for ecological assessment endpoints lower than 20% would appear to be acceptable based on current US EPA regulatory practice and could not reliably be confirmed by field studies, and can therefore be considered *de minimis* (Suter et al. 1995).
- Environment Canada (2005) advises against estimating an endpoint within the acceptable range of effect in the control(s). Beyond that point, any EC_{20}/IC_{20} would be suspect if it was below the lowest effect observed for the test concentrations. Because chronic toxicity tests typically have acceptable control responses of up to 20%, there is an increased risk of false positives when effect sizes below 20% are considered.

Consideration of Endpoint Variability

In toxicity testing, the cause of biological variability in organism responses often remains the largest source of uncertainty in controlled laboratory studies. Organism response may vary both within a test (i.e., across replicates) and between tests. This variability in test endpoints can influence the ability to detect meaningful differences (e.g., >20%) in biological responses between tested sediments. The precision of a test endpoint is commonly assessed by determining the coefficient of variation (CV) based on average responses across test replicates, but is not exclusively used because CVs can be low both in samples reflecting a wide range of toxicity (Burton et al. 1996). If CVs are high in negative controls and reference samples there is increased uncertainty regarding whether the test has the minimum detectable difference needed to indicate whether a change is actually meaningful.

Both the sensitivity to detect changes and the variability associated with a test endpoint typically depend on the type of endpoint. Sublethal endpoints such as growth of *C. dilutus* and *H. azteca* are typically more sensitive than survival as indicators of contaminant stress, but often can be more variable (US EPA 2000; Norberg-King et al. 2006). Endpoints with greater variability yield higher minimum detectable differences between control and test sediments. In the Mount Polley mine toxicity tests, growth was the more sensitive endpoint for both test species, but both endpoints tended to be highly variable depending on the sediment tested (Table 43).

In 2014, the laboratory control CV for *H. azteca* growth was high (up to 57%), which increased the uncertainty of detecting of a statistically significant or meaningful difference relative to exposed samples. The laboratory control CVs for other endpoints and in subsequent years was lower, typically <20%, which reduced uncertainty in these comparisons (Table 43). The CVs for Bootjack and Polley lake samples were typically <20% for each endpoint and species, indicative of lower uncertainty that is consistent with the limited biological response observed (i.e., no adverse effects). For the Hazeltine Channel, CVs were high for both endpoints and species and corresponded to some of the largest magnitudes of effect observed. The CVs for Quesnel Lake reference samples were high for both endpoints and species, particularly in 2014, which increased the uncertainty of detecting a meaningful difference compared to exposed samples (Table 43). The 2014 samples were tested as laboratory replicates and are expected to have lower CVs than the 2015 and 2016 samples tested as field replicates because of the potential for spatial variation in sediment chemistry and composition typically associated with field replicates. Therefore, caution should be taken in the interpretation of potential for toxic effects in exposed sediment relative to reference because as within-test variability in endpoints increases (i.e., increased CVs), particularly in the control and reference samples, the ability to determine a meaningful difference decreases and becomes more uncertain.

Table 43: Range of Coefficients of Variation (%) in Sediment Toxicity Tests

Year	Sample	10-d *Chironomus dilutus*		14-d *Hyalella azteca*	
		Survival	Growth	Survival	Growth
2014	Laboratory control	0–20	5–13	0–14	15–57
	Bootjack Lake reference	6–23	7–20	5–9	15–21
	Quesnel Lake reference	0–58	3–21	5–43	13–48
	Hazeltine Channel	15–76	14–63	6–91	16–121
	Polley Lake	10–23	11–23	5–8	18–38
	Quesnel Lake	8–70	11–45	0–71	5–74
2015	Laboratory control	0–11	7–8	6–19	8–23
	Quesnel Lake reference	5–25	9–13	0–6	25–27
	Quesnel Lake	5–13	6–14	11–23	21–33
2016[a]	Laboratory control	5	29	0	7
	Bootjack Lake reference	0	7–15	0–9	7–8
	Quesnel Lake reference	10–52	9–13	0–5	9–27
	Polley Lake	8–24	10–16	6–11	16–21
	Quesnel Lake	9–24	2–49	0–41	23–50

(a) Does not include spatial gradient or dilution series treatment because no replication was available for those samples to calculate CVs.

4.7.1.4 Confounding Factors

There are other factors that have the potential to confound the interpretation of toxicity test results, for example, overlying water quality within a test vessel can affect organism performance, which is why standard water quality parameters (e.g., pH, conductivity, ammonia) are typically measured throughout the test duration. Concentrations of ammonia in overlying water were not elevated or markedly different between control, reference, and exposed sediments. Ammonia concentrations in the TOC-supplemented treatments were higher than in the corresponding unaltered samples, but organism survival was not significantly different in these treatments[67]. Sulphide, which may also impair conditions within a test, was not measured in overlying water or porewater.

4.7.1.5 Evaluation of Causation—Physical vs Chemical Stressors

An evaluation of the potential cause of the observed toxicity was undertaken using several approaches following the 2014 toxicity testing program. The general approach entailed the following:

- post-breach tests and correlation analysis of the 2014 toxicity results with sediment chemistry
- literature review and experimental testing with additional treatments in the 2015 and 2016 programs to examine the influence of sediment physical and chemical factors
- additional correlation analyses of the 2014–2016 data to identify the relative contributions of physical and chemical factors

Results of these investigations are summarized in the following subsections.

Post-Breach Tests and Associated Correlation Analyses

Sediments from the Hazeltine Channel and profundal areas of Quesnel Lake exhibited physical differences from sediments generally present in aquatic environments and from reference areas of Quesnel Lake, and distinct from historical samples from the Hazeltine Channel. Physical differences in tailings-influenced samples included uniform particle size, relatively high density, and very low TOC content. Chemical differences included elevated metal concentrations. Minnow explored the potential relationships between sediment toxicity and physical and chemical characteristics in the 2014 data set using correlation analysis (see Appendix E of MPMC 2015 for details).

- For Polley Lake, correlation analysis indicated few significant relationships between the test endpoints and sediment characteristics.
- Survival and growth of *C. dilutus* exposed to the Hazeltine Channel sediment were weakly associated with chemical characteristics, whereas there was some association between *H. azteca* endpoints and physical and chemical characteristics (e.g., percent fines, TOC).

[67] An exception was poor survival (68%) of *H. azteca* in a supplemental control (negative control sediment with added TOC). It was noted by the laboratory that TOC did not mix well with the control, resulting in accumulation on the sediment surface and signs of fouling. Ammonia concentrations in this treatment were elevated (35 mg/L-N) and likely contributed to the poor survival based on reported mortality in this test species at concentrations above 14 mg/L-N (Borgmann 1994).

- Growth of *H. azteca* in littoral and profundal samples from Quesnel Lake tended to increase with distance from the near-field area, likely in association with increasing TOC. Survival of *C. dilutus* and survival and growth of *H. azteca* were negatively correlated with copper in littoral samples. For profundal samples, *H. azteca* endpoints were not correlated with sediment characteristics, whereas survival of *C. dilutus* was positively related to TOC and survival and growth negatively correlated with copper.

- Overall, correlation analyses indicated some consistent relationships between survival or growth and sediment physical or chemical characteristics for the Hazeltine Channel and Quesnel Lake. Specifically, there were positive relationships with TOC and negative relationships with metal concentrations, including copper. As the positive influence of TOC could not be separated from potential negative effects of elevated metals, it was uncertain whether the effects observed were due to physical (i.e., due to low TOC) or chemical factors.

Experimental Testing

Experimental testing was included in the 2015 and 2016 programs to examine the influence of particle size and TOC on toxicity test (as described in Section 4.7.1.1). Organic carbon was shown to affect survival and growth of *C. dilutus* and growth of *H. azteca* (Appendix L-4). Sediment dilution treatments were included because if significant toxicity were present and due primarily to chemical factors such as copper concentration, the toxicity would not be expected to decrease substantially from a dilution with natural sediment (which also contains some copper). However, if substrate conditions such as particle size or TOC were significant explanatory factors for the observed toxicity, the addition of natural sediment could ameliorate this response. Results of this supplemental testing program found:

- In 2015, a 50% dilution of near-field sediment reduced growth effects and slightly increased survival, which was not adversely affected. The TOC in this diluted sample would have been approximately 1.1% based on the average measured TOC in each sample. In 2016, survival of *H. azteca* was not adversely affected once the near-field sample was diluted to 60% (corresponding to ~1% TOC). Growth effects in both species were eliminated once the near-field sample was diluted to 20% (corresponding to ~1.6% TOC). This treatment was intended to be a supporting line of evidence in the exploration of potential causes for laboratory-observed toxicity and mimics a long-term field scenario in which near-field sediment is gradually altered as natural sediment processes occur through deposition and bioturbation.

- Survival of *C. dilutus* was not affected in the profundal near-field sediment and clay control samples in 2015, but was poor (58%) in the clay control in 2016. Survival of *H. azteca* was significantly reduced in the clay control in 2015 and in both near-field and clay samples in 2016. For both species, there were no large differences in growth between the near-field and clay control samples, as growth was significantly reduced relative to the laboratory control or reference (Appendix L-4 and L-5). Minnow observed that the influence of pH could not be ruled out for the clay control, as pH was low in overlying water (~pH 4.5–5.5) and the sample itself (pH 4.5). However, pH was similar in the overlying water of the clay control samples with and without TOC.

- For both species, the significant effects on growth observed in near-field sediment were eliminated with the addition of TOC. Growth in the other TOC-supplemented sediments typically was greater than in the corresponding unaltered samples. The adverse effects observed for *H. azteca* in the clay control in 2015 were also reduced with the addition of TOC. (Appendix L-5).

- A literature review was conducted to evaluate the potential for confounding effects of substrate type on the results of *C. dilutus* and *H. azteca* toxicity tests. Overall, the review indicated that both species generally tolerate a wide range in sediment particle sizes, although some particle-size effects have been observed for *C. dilutus* survival and growth. Results from these experimental treatments are consistent with literature-reported findings that *C. dilutus* is intolerant of sediment with TOC lower than approximately 1% (Suedel and Rodgers 1994) and that larval growth is not strongly related to grain size (Sibley et al. 1997). However, both survival and growth of *H. azteca* appeared to be influenced by TOC based on the reduction of effects when TOC was added to the clay control. The TOC content was low in both the near-field sediment (0.35%) and the clay control (0.8%). The consistent positive response to TOC amendment in both near-field sediment and clay control (uncontaminated but with high fines content) suggests that the response to TOC was through provision of food resources rather than through additional metal binding capacity. This strongly indicates that low TOC is a likely cause of growth effects in *C. dilutus* in exposed samples.

2014 to 2016 Correlation Analysis

A correlation analysis was undertaken by Golder using the 2014–2016 data set to identify the relative contribution of physical and chemical stressors to the observed toxicological responses (Appendix L-6). Methods of data analysis included:

- The 2014–2016 toxicity results and corresponding sample-specific sediment chemistry were compiled into one data set, but excluded data from the 28-day tests, diluted samples, clay controls, TOC-supplemented samples, and any samples without paired sediment chemistry (as provided by Minnow). Growth endpoints were normalized to the mean response in the paired negative laboratory control. Survival was not normalized.
- Sediment metal chemistry data were summarized using a Principal Components Analysis (PCA; Systat v13). Analytes with incomplete data sets were excluded from the data matrix prior to analysis. All sediment chemistry data were transformed using log(concentration x + 1), as concentration data are generally log-normally distributed. PCA axes (PC1 and PC2) were generated for the following data sets: all areas, lakes only, Quesnel Lake, and Polley and Bootjack lakes. Correlation analysis was conducted between PC axes and metals.
- Spearman rank correlation analysis was conducted between toxicity test endpoints and the following: PC1 and PC2, arsenic, copper, particle size (% gravel, % sand, % silt, % clay), and TOC. Significant correlations (p <0.01) and strong correlations (R^2 = 0.5 for Quesnel Lake and 0.26 for Polley and Bootjack Lakes) were identified. Due to large sample size for Quesnel Lake, many of the parameters were significant and an R^2 value was selected to represent strong correlations to focus on key parameters.
- The relationship between toxicity endpoint, copper, and TOC was examined by combining data for all areas, as this provided a larger data set that was no longer influenced by lake-specific geochemistry. Visual examination with 3-dimensional graphs did not show a response that follows a predictive relationship based on the current understanding of the mechanism of copper toxicity. Instead, data were binned into three ranges of TOC values (0.1–1%, 1–10%, 10–20%) and shown on an overlapping scatterplot (Figure 52).

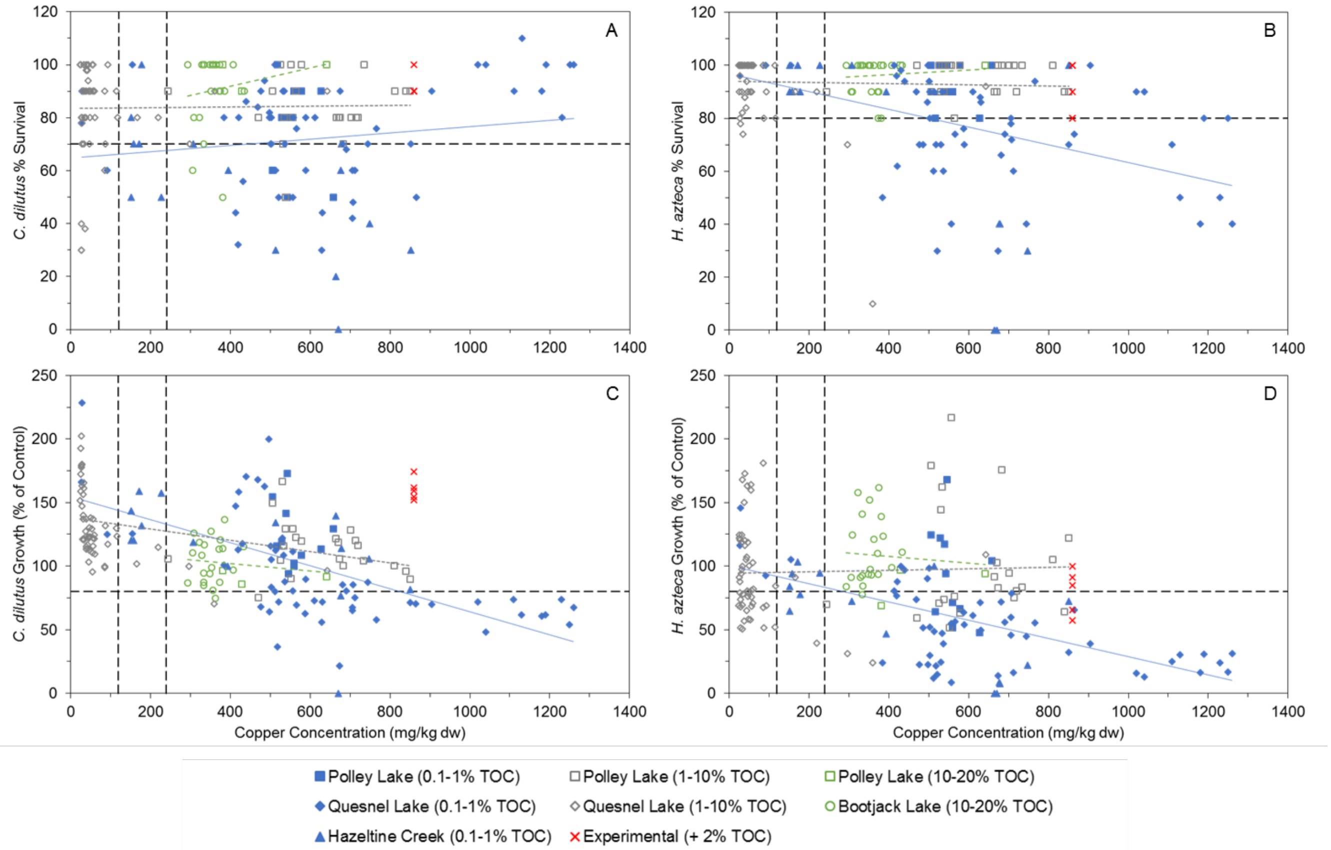

Figure 52: Response in 10-Day Chironomus dilutus and 14-Day Hyalella azteca Toxicity Tests as a Function of Sediment Copper Concentrations and Degree of Total Organic Carbon Amendment

Notes:

Blue solid line—linear relationship for 0.1–1% TOC range.

Grey dotted line—linear relationship for 1–10% TOC range (experimental +2% TOC not included).

Green dashed line—linear relationship for 10–20% TOC range.

Vertical dashed lines—$SedQC_{SS}$ (120 mg/kg copper) and $SedQC_{TS}$ (240 mg/kg).

Horizontal dashed lines—protocol-specified performance criteria for survival and growth of controls.

The key findings from the multivariate analysis and Figure 52 were:

- PC1 and PC2 explained approximately 62% of the variance for all areas or lakes, but improved when the analysis was separated (72% for Quesnel Lake; 77% for the combined and Polley and Bootjack Lakes). Golder infers that these differences relate to different geochemistry between lakes (particularly Bootjack Lake) that would overwhelm the ability of the PCA to detect the influence of any particular parameter. Subsequent analyses were separated by lake to distinguish the main factors that may be influencing toxicity responses.

- Spearman rank correlation analysis showed that toxicity responses in Quesnel Lake samples were most strongly correlated with copper (negative correlation) and TOC (positive correlation) and were significant for most endpoints. For Polley and Bootjack Lakes, TOC was the only parameter that was significantly correlated across multiple toxicity endpoints, but was not consistently positive or negative for *C. dilutus* endpoints.
- In terms of different endpoints:
  - There was no relationship between copper concentration and survival of *C. dilutus*, whereas there was a negative relationship with *H. azteca* survival for samples with TOC <1% (Figure 52 A and B). Survival was not adversely affected in the majority of samples.
  - There was a negative relationship between copper and growth in both species for samples with TOC <1% (Figure 52 C and D). Effects were not as prominent in *C. dilutus* because the growth endpoint was not reduced by more than 20% relative to the control (Figure 52) or reference (Figure 50) in the majority of samples.
  - *Hyalella azteca* showed greater effects, as a larger proportion of samples exhibited a >20% reduction in growth relative to the control, even in samples with low copper and TOC >1%. Visual examination of all toxicity data showed that at similar sediment copper concentrations, toxicity typically was not observed in samples with TOC >1%. This ameliorating effect of TOC is supported by the experimental treatments in which toxicity was reduced when 2% TOC was added to exposed sediments with elevated concentrations of copper.

4.7.1.6 Uncertainty Analysis

Standardized test methods and validation criteria, including quality QA/QC practices, were applied to laboratory toxicity tests to confirm data reliability and repeatability. The selected test species (i.e., *C. dilutus* and *H. azteca*) are sensitive, widely-used test species for which standardized testing procedures are available. Both test species are considered representative of the genera present in most of the management areas. The selected standard protocols for both species satisfy the CCME (2007) definition of a long term test. Inclusion of a longer test with *H. azteca* did not show a meaningfully increased sensitivity in response relative to the standard test duration.

Physical conditions, such as organic carbon and grain size distributions, can confound determination of causal factors of toxicity. Clean sediments with a range of physical characteristics were included in the testing program, along with TOC supplementation as additional treatment, to help understand the relative influence of physical factors and residual metal concentrations on toxicity test results. These added controls and treatments suggested that lack of TOC and fine particle size likely contributed to the effects observed. It was not possible to use the combined toxicological data set (i.e., un-modified, field collected samples and supplemental controls or treatments where negative control sediment or field collected samples were modified by the addition of organic carbon) to develop a model that could quantitatively predict the cumulative influence of copper and TOC on test performance. This uncertainty may be partially due to issues related to how organic carbon was added to the experimental system (i.e., as noted in footnote 67, there was an instance where clean sediment supplemented with organic carbon resulted in unacceptable performance). However, the uncertainty in developing a predictive multi-variate relationship is more likely influenced by the fact that copper and organic carbon are not independent variables nor do they demonstrate a consistent relationship to survival or growth along the gradient. Rather, the literature demonstrates at least three different types of interaction:

- In samples with high organic carbon, the organic carbon provides an excess of food for test organisms and can also act independently to sorb copper.
- In samples with moderate organic carbon, there is sufficient food for organisms, but the binding capacity would be diminished.
- In samples with low organic carbon, organisms can be stressed by insufficient food, and the binding capacity of copper to organic carbon would be further diminished.

The statistical evaluation of data did not have sufficient resolution to determine the threshold points between these types of interaction. Ultimately, it is not necessary to pursue a mechanistic or mathematical model beyond the information that has been presented in Figure 52 – the existing data shows that test performance can be expected to improve over time as the natural organic carbon concentrations are reestablished.

It is possible that laboratory-reared test organisms may be more sensitive to contaminants in sediment than the actual organisms at the site. Field organisms are acclimated to local conditions and develop tolerance to concentrations of metals that may impact laboratory-reared biota. The toxicity test is conducted under laboratory conditions that cannot fully capture all the natural variables that can influence bioavailability and how the broader benthic community responds to those metals. However, laboratory testing can evaluate potential effects to organisms in cases where the benthic community may not be present due to physical disturbance and the time required for recolonization to occur, as was the case for benthic sampling immediately after the TSF embankment breach.

Toxicity testing allows for a direct measurement of survival and growth, but is not necessarily representative of field conditions. For example, in spite of the apparent magnitude of response in toxicity testing with *H. azteca* in Quesnel Lake littoral areas, there were several hundred *Hyalella* per square meter identified in LFF and LREF-1 benthic taxonomy samples from these areas, suggesting that toxicity may not be indicative of population responses. Therefore, the strength of association to the assessment endpoint is considered moderate.

4.7.1.7 Comparison to Decision Criteria

Decision criteria were established based on no more than a 20% reduction in endpoints relative to reference samples.

<table>
<tr><th>Line of Evidence</th><th>Assessment Endpoint</th><th>Measurement Endpoint(s)</th><th>Decision Criterion</th><th>Area</th><th>Decision</th></tr>
<tr><td rowspan="3">Sediment toxicity</td><td rowspan="3">Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings.</td><td rowspan="3">Conduct 14-d amphipod survival and growth and 10-d chironomid survival and growth toxicity tests on field collected samples.</td><td rowspan="3">Toxicological performance is not reduced by more than 20% relative to reference samples.</td><td>Polley Lake</td><td>No significant toxicity</td></tr>
<tr><td>Hazeltine Channel</td><td rowspan="2">Potential risk, but likely influenced by physical factors, particularly TOC</td></tr>
<tr><td>Quesnel Lake littoral and profundal</td></tr>
</table>

The totality of the available toxicity data show that reductions in endpoint performance cannot be attributed to copper alone as low organic carbon is also strongly influencing test performance. The influence of low TOC is interrelated with copper because the presence of low TOC as well as copper co-occur in the deposited tailings. There is a relationship that shows adverse effects in low TOC sediments are likely related to the copper concentration, but both the multivariate analysis and experimental manipulation showed that the toxicity will be substantially reduced by the presence of TOC. Therefore, it is expected that the natural accumulation of TOC in Quesnel Lake will ameliorate the temporary effects that may currently be attributed to copper.

4.7.2 Benthic Community Structure

4.7.2.1 Study Design and Methodology

Benthic invertebrate community samples were collected by Minnow from a number of locations over three years as part of the PEEIAR process. Benthic community data were collected to evaluate comparisons between:

- Upper Hazeltine Channel and lower Edney Creek (exposed areas) and a reference area (upper Edney Creek) based on samples collected in 2015. Edney Creek was also resampled in 2016. Data were also available from previous sampling conducted in 2007 from upper Hazeltine Channel.
- Quesnel River (potentially exposed area) and a reference area (Cariboo River) based on samples collected in 2014 and 2015.
- Polley Lake (exposed area) and a reference area (Bootjack Lake) based on samples collected in 2014, 2015 and 2016.
- Quesnel Lake (shallow) and Quensel Lake (deep) (exposed areas) versus reference areas elsewhere in Quesnel Lake based on samples collected in 2014, 2015, and 2016.

Each of these sampling programs was intended to facilitate pair-wise comparisons (e.g., between representative locations from exposed and reference area, or between years at the same location). Data were not collected as part of a gradient design. A detailed methodology associated with sample collection, laboratory analyses, data management, data assessment, and data interpretation are provided in Appendix O-1. Benthic invertebrate samples were collected following standardized methods that were consistent with established provincial and federal protocols (i.e., BC MoE 2013; Environment Canada 2012a). Specifically:

- Quesnel Lake, Polley Lake, and Bootjack Lake—At each station, a petite ponar grab was used to collect benthic invertebrate samples. Five subsamples were collected and combined into a composite sample at each station; each composite sample was sieved through a 250-µm mesh sieve bag in the field.
- Hazeltine and Edney Creeks—No benthic invertebrate community sampling was conducted in the Hazeltine Channel in 2014 due to the absence of appropriate erosional habitat following the tailings spill (i.e., substrates were entirely fine materials derived from the tailings and scoured creek bed). Benthic invertebrate communities of the Hazeltine Channel and Edney Creek (reference) were characterized in subsequent years using a combination of Surber sampling and kick-net sampling. Surber samples (500-µm mesh) was used to collect three subsamples at each station which were combined in into a single composite sample. Additional benthic invertebrate community samples (one replicate per station) were collected from creeks using a kick-net sampler equipped with 400-µm mesh under the Canadian Aquatic Biomonitoring Network (CABIN) sampling protocol.

- Quesnel and Caribou River—Benthic invertebrate community samples were collected from the shallow margins of the Quesnel and Cariboo (reference) rivers using the CABIN protocols.

4.7.2.2 Summary of Pairwise Comparisons

Appendix O provides information regarding benthic community structure from Minnow. These files included:

- A detailed discussion regarding the relative proportions of different taxonomic groups in each contrast (exposed versus reference area) based on the data collected in 2014 and 2015.
- Statistical comparisons for each pair of exposed and reference locations for each year. A detailed discussion of these data from 2014 and 2015 is also provided, and an updated summary table was provided with the results of the pair-wise comparisons for 2016 for Polley and Quesnel Lakes.
- A summary table was also provided by Minnow for pair-wise comparisons within a location between different years (2016 versus 2015; 2016 versus 2014).

Pair-wise comparisons were conducted for a wide variety of metrics, including density, richness, Simpson's Diversity Index, Simpson's Evenness Index and Bray-Curtis Index. Pair-wise comparisons were also made based on the output of the statistical reduction of the taxonomy data (i.e., correspondence analysis axis 1 based on 2016 data, as well as non-metric multidimensional scaling axis 1 for 2014 and 2016 data.). These last two techniques are intended to reduce the totality of the benthic data to a single variable. Additional pairwise comparisons were made on the densities and percentages of major taxonomic groups (Appendix O-1). Broadly, the results from these exploratory pairwise comparisons in 2014 and 2015 were used by Minnow to refine the benthic community sampling program for 2016; these analyses have also been used by Golder to help to focus the risk assessment evaluation of the benthic community data.

Minnow concluded that there were no meaningful differences between the Quesnel River location and its reference station based on the sampling completed in 2014 and 2015. (Appendix O-1, page 12). The mean taxa richness, the two Simpson index values, and the correspondence analysis axis 1 were all consistent between areas and years. There were differences in benthic abundance between the Quesnel River and its reference location for one year, but not the other, which was considered by Minnow to be the result of natural variability.

4.7.2.3 Summary of Comparison to Risk Decision Criteria

Benthic community data were evaluated using a decision criterion with a 20% level of response. Specifically, data would not be considered indicative of an adverse effect unless there was at least a 20% (and statistically significant) reduction in endpoint performance relative to the reference area (matched to year of collection). The use of a 20% decision criterion (often in conjunction with statistical differences) is consistent with common risk assessment practice for contaminated sites in British Columbia, as discussed in the context of the toxicity testing analysis in Section 4.7.1.

Two benthic community metrics were selected for evaluation against the risk-based decision criterion:

- **Total abundance (density)**—Absolute number of individuals captured as mean density (organisms/m^2). This metric directly evaluates the productivity of the benthic community as a source of fish food organisms.

- **Simpson's diversity index (SDI)**—The SDI measures the proportional distribution of organisms in the community, taking into account the taxonomic richness and how evenly the density is distributed among these taxa. The SDI is effectively a measure of "dissimilarity" between locations. SDI values range between zero and one; a lower value indicates a less diverse or evenly distributed community. This metric provides information about the overall "health" of the benthic community.

Other mathematical calculations for "dissimilarity" are available (e.g., Bray-Curtis; Shannon diversity index) that would use the same underlying data regarding diversity and abundance in the samples. Statistical reduction techniques that reduce complex data sets (such as taxonomy) to one or two axis are also available. There are also multiple indices that can be applied to represent the underlying benthic community and many purport to provide a defensible relationship between the cumulative effect of different stressors and the "health" of the benthic community (for example, the index of biotic integrity; IBI). Numerous examples of IBI-style indices exist in the literature, but all require information about the variability inherent in the unimpacted reference condition, as well as the ability to determine how the magnitude of effects change over the gradient of the specific stressors being measured. Ultimately, total abundance and SDI were selected as representative functional measures of community health; these metrics are intuitive and provide the least degree of transformation or manipulation from the source data.

Table 44 presents the results of the selected metrics for samples collected from the Hazeltine Channel, Polley Lake and Quesnel Lake in the context of the decision criteria:

- **Polley Lake (Deep)**—Both deep locations in Polley Lake (POL-P2 and POL-P1) exhibited a greater than 20% reduction and statistically significant reduction in mean density and diversity in the first year of sampling. However, there are clear signs of recovery in subsequent monitoring. Density in POL-P2 is no longer reduced by more than 20%, but the diversity of the benthic community has not yet recovered. Density in POL-P1 is still reduced (but is not statistically significant) whereas diversity has recovered.
- **Polley Lake (Mid-depth)**—Both mid-depth locations in Polley (POL-2 and POL-1) exhibited a greater than 20% reduction and statistically significant reduction in mean density in the first year of sampling. Diversity was also reduced by more than 20%, but the reduction was not statistically significant. As with profundal Polley Lake, there are clear signs of recovery. Density in 2016 was higher than reference. Diversity in 2016 was reduced by 8% at POL-1, and by 31% (not statistically different) at POL-2.
- **Quesnel Lake (Deep)**—Both PNF (near-field) and PFF1 (far-field) has exhibited a statistically significant and greater than 20% reduction in mean invertebrate density in all three sampling years. No statistically significant and greater than 20% reductions in mean SDI have been observed, but overall, recovery at these stations is slower than what has been observed in Polley Lake. This is not surprising given that Quesnel Lake constitutes a much larger area that will naturally take longer for recolonization from unimpacted areas along the margins.
- **Quesnel Lake (Shallow)**—The far-field station (LFF) has never shown reductions in density or diversity. However, the near-field station (LNF) continue to exhibit a statistically significant reduction greater than 20% in mean density and SDI and recovery is not yet evident.
- **Hazeltine Channel**—There were statistically significant reductions (greater than 20% reduction) in diversity at HAC-U (upstream) and HAC-D (downstream) compared to EDC-D, irrespective of whether data were collected with a Surber sampler or a kicknet. Data was limited to 2015, and creek rehabilitation was subsequently initiated. No further sampling has yet been conducted.

- **Lower Edney Creek**—Similar to the Hazeltine Channel, sampling has been conducted with both a Surber sampler and a kick-net. Storey et al. (1991) generally found that Surber samplers would collect fewer individuals but more taxa than a kicknet because the Surber sampler has a lower surface area but more intense sampling method. Differences in methodology introduce additional uncertainty into the risk conclusions. Broadly, the combined data show that there were statistically significant reductions (greater than 20% reductions) in both density and diversity in 2015, but that conditions have improved in 2016.

4.7.2.4 Exploratory Multivariate Analysis

Minnow completed Spearman's Rank correlations between the array of benthic community metrics[68] and a variety of exposure metrics including physical variables (e.g., depth), water quality variables, concentration of individual metals in sediment, and multivariate reduction axes (e.g., PC1 for metals in sediment) based on the data from 2014 and 2015.

The exploratory correlation analyses, although an important element for developing a meaningful long-term monitoring program, are not directly relevant to the objectives of the risk assessment. An exploratory analysis that identifies a statistically significant difference in one metric/variable pair between two areas is not necessarily indicative of an ecological relevant effect or an underlying cause-effect relationship related to the deposition of tailings in general. This challenge exists even if the statistical analysis compensates for the bias of making multiple comparisons with the same data set[69] because an exploratory statistical analysis has not attempted to identify the causal factor(s) or specific measure(s) of effect that provides the most information relevant to the assessment endpoint. Findings from the various exploratory statistical analysis have not been incorporated in the benthic community line of evidence, but will be used to help guide long-term monitoring where appropriate.

If risks are found to be unacceptable, there is value in identifying a small number of specific metrics that can be shown to be adequately responsive to long-term trends in benthic community performance without being overwhelmed by year-to-year variation associated with natural biological variability or other sources of variance. In these circumstances, the appropriate decision is to focus on the metric that provides the closest strength of association with the assessment endpoint—namely, the underlying abundance and diversity data that is used to calculate all the various subsidiary metrics.

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[68] As noted in previous sections, benthic community metrics included density, various similarity and diversity indices, multivariate data reduction axes and density and percentages of selected taxonomic groups. The total number of benthic community metrics varied from approximately 15 to 25, depending on the waterbody. See Appendix E of the Minnow report included in Appendix O of this current risk assessment report.

[69] Minnow applied Tukey's method or Dunn's method with a Bonferroni correction on alpha (see discussion in Appendix A of Minnow's report included as Appendix O of this current risk assessment). The purpose of this adjustment is deal with the fact that a statistically significant pair-wise comparison will be identified at some point by the sheer number of comparisons being made.

Table 44: Results of the 2014, 2015 and 2016 Weight of Evidence Samples for Benthic Taxonomy

Exposure Area	Year	Exposure vs Reference	Mean Density (organisms/m^2)		Percent Reduction	Mean SDI		Percent Reduction
			Exposure	Reference		Exposure	Reference	
Polley Lake (Deep)	2014	POL-P2 vs BOL-B2	0	1,862	**100%**	NC	0.24	**100%**
	2015		336	1,448	**76.8%**	0.43	0.63	32%
	2016		1,360	1,245	No reduction	0.41	0.71	**42%**
	2014	POL-P1 vs BOL-B2	11	1,862	**99%**	0.17	0.24	29%
	2015		NS	1,448	NC	NS	0.63	NC
	2016		686	1,245	45%	0.57	0.71	20%
Polley Lake (Mid-depth)	2014	POL-2 vs BOL-2	136	1,534	**91%**	0.33	0.50	35%
	2016		4,228	1,359	No reduction	0.47	0.68	31%
	2014	POL-1 vs BOL-1	93	1,534	**94%**	0.48	0.50	5%
	2016		13,005	1,359	No reduction	0.62	0.68	8%
Quesnel Lake (Deep)	2014	PNF vs PREF1	40	5,816	**99%**	0.39	0.38	No reduction
	2015		216	17,293	**99%**	0.67	0.28	No reduction
	2016		588	5,690	**90%**	0.44	0.53	17%
	2014	PFF1 vs PREF1	22	5,816	**100%**	0.54	0.38	No reduction
	2015		NS	17,293	**NC**	NC	0.28	NC
	2016		266	5,690	**95%**	0.47	0.53	13%
Quesnel Lake (Shallow)	2014	LFF vs LREF1	5,402	1,421	No reduction	0.87	0.85	No reduction
	2015		5,943	9,652	38%	0.89	0.78	No reduction
	2016		5,472	4,424	No reduction	0.84	0.82	No reduction
	2014	LNF1 vs LREF1	159	1,421	**89%**	0.74	0.85	13%
	2015		NS	9,652	NC	NC	0.78	NC
	2016		59	4,424	**99%**	0.32	0.82	**61%**

Exposure Area	Year	Exposure vs Reference	Mean Density (organisms/m^2)		Percent Reduction	Mean SDI		Percent Reduction
			Exposure	Reference		Exposure	Reference	
Hazeltine Channel	2015	HAC-U vs EDC-D (Surber)	28,143	5,710	No reduction	0.51	0.91	**45%**
	2015	HAC-D vs EDC-D (Surber)	2,992	5,710	49%	0.66	0.91	**28%**
	2015	HAC-U vs EDC-D (CABIN)	64,300	11,963	No reduction	0.55	0.91	**40%**
	2015	HAC-D vs EDC-D (CABIN)	10,169	11,963	15%	0.08	0.91	**91%**
Lower Edney Creek	2015	E1 vs EDC-D (Surber)	3,532	5,710	38%	0.52	0.91	**44%**
	2016		6,383	3,867	No reduction	0.86	0.91	6%
	2015	E2 vs EDC-D (Surber)	4,968	5,710	13%	0.73	0.91	20%
	2015	E1 vs EDC-D (CABIN)	5,185	11,963	**57%**	0.59	0.91	**36%**
	2016		2,691	2,765	3%	0.91	0.93	2%
	2015	E2 vs EDC-D (CABIN)	10,906	11,963	9%	0.76	0.91	16%

Bold highlight indicates greater than 20% reduction that is statistically signficant. NS = Not sampled; NC = Not calculated.

4.7.2.5 Recolonization Studies

Site-Specific Deployments

A sediment transplant study was completed by Minnow in 2015 to determine if sediment stressors (either physical or chemical) were likely to present a barrier to successful recolonization. A detailed description is provided in Appendix O-1, but in brief:

- Bulk sediment samples were collected from an exposed area (PNF) and a reference area (PREF-1) and homogenized. The mean concentrations of the COPCs and primary physical variables were:
  - Exposed PNF sediments—Copper (859 mg/kg versus a $SedQC_{SS}$ of 120 mg/kg); arsenic (15 mg/kg versus a $SedQC_{SS}$ of 11 mg/kg); total organic carbon (0.3%); 93% percent fines. No other substances exceeded the numerical standards. This sample was considered representative of strongly tailings-influenced sediments.
  - Amended—This was the same material as PNF, but supplemented with a mix of peat and Tetramin fish food to achieve a nominal total organic carbon of 2%.
  - Reference—Copper 55 mg/kg, arsenic 8.9 mg/kg, total organic carbon 1.8%, 80% percent fines. This sample was considered representative of Quesnel Lake without tailings.
- Sediment was placed into buckets with a depth of 10 cm and an approximate area of 350 cm^2. Organisms in the original sediment were not removed and would have been included in the homogenization and distribution of sediment to the replicate buckets. These buckets were then placed back into each of the two locations in a randomized design. A total of six replicates of each treatment was placed in each area. Note that this included the original sediment placed back into the same source area to act as a bucket control.
- Buckets were left in place for approximately seven weeks and then the contents were passed through a 250 micron sieve. The material retained on the sieve was submitted for taxonomic analysis. Grab samples were also collected from each area before and after deployment and submitted for taxonomic analysis.

A graphical representation of the study design is provided in Figure 53. As noted in Section 4.7.2.1, Minnow calculated a wide array of summary metrics, pair-wise comparisons, and exploratory statistical analysis, which can be found in Appendix O. A summary of the density and Simpson's diversity values in each treatment is provided below in Figure 54.

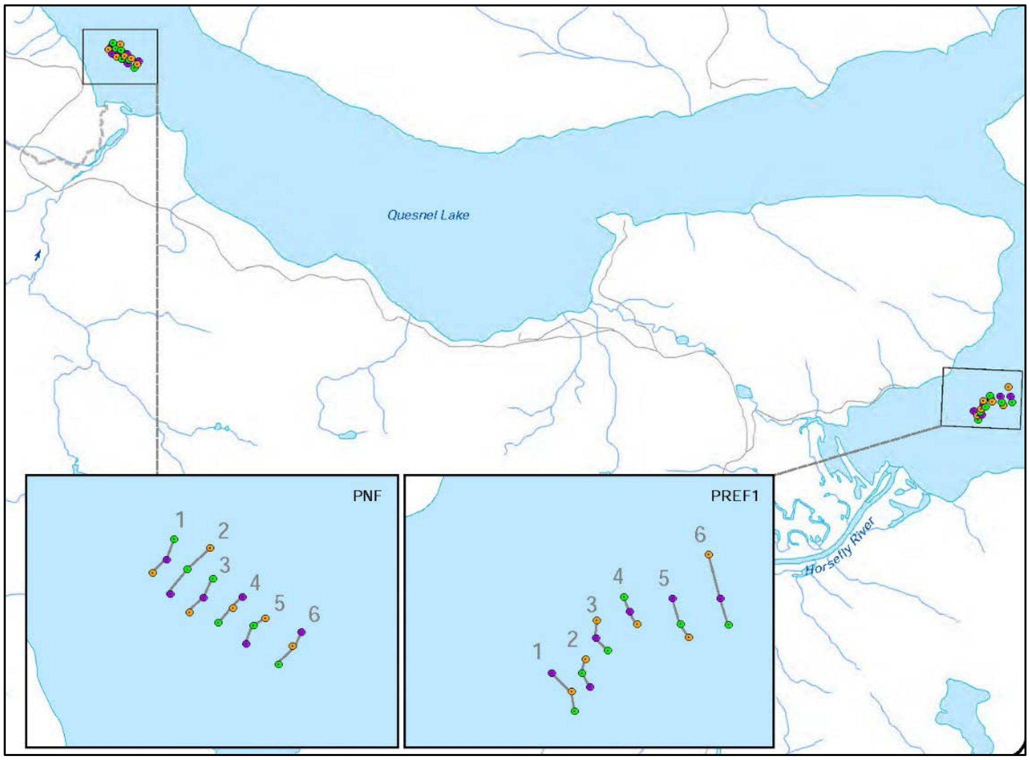

Figure 53: Recolonization Study Design (from Minnow 2016; see Appendix O)

Note: Green symbols = reference sediment; magenta symbols = exposed sediment; orange symbols = amended sediment.

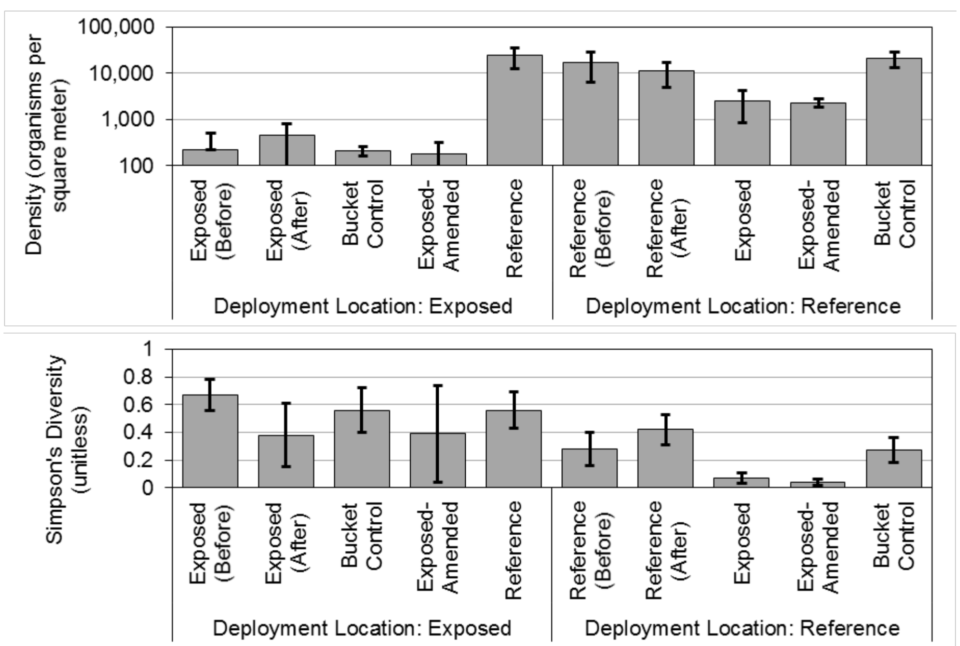

Figure 54: Density and Simpson's Diversity Index Values (Mean with Standard Deviation) for Treatments in the Benthic Recolonization Study

The transplant study showed that:

- The low organic carbon and elevated metals in the exposure sediment was not a barrier to recolonization. Both the exposed and amended material were colonized by chironomids (two taxa), water mite (one taxon) and oligochaete worms (one taxon). Chironomids tend to be an early colonizing species (see discussion of Fairchild et al. 2012, below). The other colonizing taxa appear to have dispersal attributes that allowed them to rapidly occupy the available sediment within a relatively short period of time.
- There was no sign that the buckets themselves would negatively influence the existing diversity and abundance in a given treatment, however, the study design itself (i.e., buckets sitting on the sediment surface) may have presented a physical barrier to some burrowing or epibenthic organisms. This is shown by the relatively low diversity index values (and the lower density values) in the exposed samples placed at the reference site. The final density and diversity of the exposure sediment if the transplant buckets were left in place for an extended duration is unknown.
- The addition of organic carbon did not result in an appreciable improvement of this early phase of recolonization. It is not clear whether this is simply because the early colonizing taxa are not sensitive to this variable, or if other variables such as grain size are playing a more substantive role.
- Near-bottom water quality at the exposure site did not exert a significant effect on the organisms in the reference sediment replicates that were transplanted to the exposure site.

Supporting Literature

A literature review was conducted to identify research projects that examined benthic community recolonization under similar circumstances. The review focused on freshwater lentic cold-water environments (e.g., marine and tropical environments were excluded) that had been subject to a large-scale chemical or physical stressor. The objective of the review was not to provide an inference regarding the long-term success or timelines involved with recolonization in the study area, but to highlight how different physical, chemical and biological variables influence this important ecological process.

Acidified Lakes – Sudbury, Ontario

Szkokan-Emilson et al. (2010) examined the benthic recolonization of acid-damaged lakes near Subury, Ontario and found that some alterations in benthic community structure persisted over the long-term, relative to a reference envelope of 20 lakes. However, the ecological relevance of the change in benthic community structure is less clear. For example, measures of species richness, Simpson's diversity, and percent EPT taxa, considered strong functional indicators of community health were within the range of reference. However, when a total of 11 different benthic community metrics were combined statistically into a multivariate analysis, all three impacted lakes were found to be significantly different than the reference condition. Further examination of the specific taxa that were contributing to the apparent differences found that some taxa were present simply because some lakes were fed by a river that acted as a source of colonists. There was also differences in the rate of colonization of different mayfly species—minor differences in pH tolerance provided an early colonization advantage for some species that then persisted as a longer term influence on benthic structure.

This study highlights two findings of potential relevance to the recolonization of aquatic habitats impacted by the TSF breach:

- the rate of recolonization will vary greatly depending on the interaction of proximity to colonist sources, dispersion potential and tolerance of each species to the physical and chemical conditions
- A long term outcome can include statistically significant differences in assemblages long after disturbance even if no functional difference to ecology is present.

Deposition of Slag, Upper Columbia River

Fairchild et al. (2012) reported on a recolonization experiment conducted with a variety of different sediment samples collected from the Upper Columbia River downstream of the Trail Smelter. Samples had a range of metals concentrations that exceeded probable effects thresholds adopted from the literature, and were also tested with a 28-d amphipod (*Hyalella azteca*) survival, growth and reproduction toxicity test, as well as a 10-d chironomid (*Chironomus dilutus*) survival and growth toxicity test. Reference sediments and amended sediments (i.e., metal-impacted sediments spiked with absorptive resin) were also included. The recolonization experiment involved deployment of 10 replicates trays of each sediment (15 cm wide by 12 cm long by 3 cm deep) into a natural pond at a depth of 1.5 meters. The total length of deployment was eight weeks. A summary of the available data indicates that:

- Concentrations of metals (including copper) that exceed literature values do not necessarily exert negative effects in standardized toxicity tests, even when the concentrations of metals are expressed as the fraction that is more likely to be bioavailable. Adverse effects in Fairchild et al. (2012) were only present in the 28-day *Hyalella azteca* test, and only at very high metal concentrations.
- Recolonization occurred with roughly the same success across all treatments, even in those treatments that had little organic carbon and that were dominated by a single grain size type.
- In recolonization treatments, there were no statistically differences amongst treatments with respect to the total number of invertebrates. There were differences with respect to the total number of organisms in the two major taxonomic groups (Chironomidae and Certapognonidae) but these differences were not related to metal concentrations. From a practical perspective, statistical differences for two groups of midges may have little ecological relevance, especially when the total number of organisms was consistent.

This study highlighted that adverse effects in a specific chronic toxicity test (such as the 28-day *H. azteca* survival and growth test) were not necessarily a reliable predictor of the success of the early stage of colonization. Fairchild et al (2012) also showed that chironomids tend to be part of the early colonizing group.

Table 45: Summary of Findings from Fairchild et al. (2012)

		DME2 (High Slag)	DME1 (High Slag)	CHNB (Medium Slag)	MFE (Low Slag)	CERC (Naural Reference)	FLSNT (Negative Control)
Grain Size	%sand	100	100	72	52	9	9
Grain Size	%fines	0	0	22	39	87	91
TOC	%	0.02	0.01	1.2	2.3	1.8	0.95
Bulk Metals Concentration	∑PEQ	132	67	11	<1	<1	<1
AVS-SEM Metals	∑PEQ	54	42	8	1	<1	<1
Amphipod Survival	%	20	92	100	98	88	88
Amphipod Growth	mg biomass	0.18	2.77	2.86	2.83	4.04	3.13
Chironomid Survival	%	95	98	100	100	100	98
Chironomid Growth	mg biomass	4.04	5.39	4.8	5.91	8.9	5.8
Abundance	Mean N of organisms	70	63	89	93	83	111
Diversity	Mean N of taxa	12	10	13	14	11	25

∑PEQ = the sum of the ratio of the concentration of Cu, Cd, Pb and Zn to the probable effect level. The sum was based on the bulk metal concentration or the extractable metals concentration depending on the metric.

4.7.2.6 Relative Influence of Chemical and Physical Stressor

One of the major objectives of this risk assessment is to separate (where possible) the influence of toxicological effects from metals from responses related to the physical nature of the deposited tailings. Toxicological effects, if present, would influence the ability of specific species to recolonize the disturbed area based on their toxicological sensitivity as well as its ability to acclimate and adapt. These responses, if present, could require a long time for recovery, relying on long-term chemical transformation processes or sediment mixing to ameliorate the toxicological responses. In contrast, physical constraints to recolonization may require shorter periods to be addressed, as physical conditions amenable to successful recolonization would occur through natural physical and biological succession processes.

Broadly, the available benthic community data provides qualitative information about the relative influence of the physical and chemical stressors:

- Minnow's exploratory statistical analysis based on the 2014 and 2015 benthic community data did not find any statistically-significant[70] and ecologically-meaningful correlations for Polley Lake[71], Quesnel Lake littoral or Quensel Lake profundal areas between the benthic community metrics and the main chemical and physical variables (copper, arsenic, percent fines, percent TOC). Although there are limitations to such exploratory statistics (Section 4.7.2.4), the absence of any statistically significant relationships highlights that a single universally applicable stressor-response relationship is unlikely to explain the patterns of biological communities.

[70] The test for significant significance requires consideration of the Bonferroni correction for the number of comparisons made, as well as the fact that there was ultimately a relatively low number of independent samples available (e.g., four or five samples per year for a total N of between 8 and 10). Minnow reported "statistical significance" with no correction for the number of comparisons made, as well as a limited correction only for the number of independent variables. Neither of these alternative presentations were considered to avoid the possibility of drawing an incorrect conclusion about a cause-effect relationship based on a potentially spurious correlation.

[71] There was a statistically significant correlation between the density of Chironomidae and arsenic concentrations in sediment, but it was a positive correlation (i.e., increasing arsenic concentration resulted in increased number of chironomids).

- There are different patterns in recolonization between Polley and Quesnel Lakes despite similar concentrations of copper in sediment. Specifically:
  - The average copper concentrations in the deep stations in Polley Lake (POL-P1 and POL-P2) ranged from 594 to 823 mg/kg, whereas the average concentrations in the deep station in Quensel Lake (PNF and PFF1) ranged from 481 to 1,210 mg/kg.
  - The average total organic carbon measurements in the same Polley Lake stations ranged from 5.0 to 9.2%, relative to a range of between 0.2 and 0.4% for the Quensel Lake stations.
  - Despite the similar range in copper exposure, Polley Lake appears to be recovering faster than Quesnel Lake. This suggests that all things being equal, it is less likely that metals are the major driver influencing the community structure, and more likely that substrate characteristics are important influences.

From a practical perspective, it is useful to recognize that the initial physical effect (smothering) associated with the initial deposition of tailings was very large, and effectively resulted in community removal. Therefore, the response to the initial physical stress is not a recovery to the original condition, but rather a succession from a new set of environmental conditions. Wood et al. (2005) showed that the ability of four different macroinvertebrate species[72] to excavate themselves from deposited material varied and was ultimately dependent on the depth and grain size of the deposited material. Some taxa could not excavate themselves from the maximum depth tested (10 mm). In the case of the TSF embankment breach, large areas of the receiving environment experiences multiple tens of centimeters (and in some cases, several meters) of physical debris, which would have resulted in initial community removal. Recolonization of a sediment with low total organic carbon and elevated concentrations of arsenic and copper has been shown to be occurring, however, the main taxa recolonizing the material was Chironomidae, which are an early successional taxon in a highly disturbed environment. Chironomidae are known to be generally able to inhabit sediments that are dominated by fines and have low organic carbon. Chironomidae were also naturally the dominant taxa in the environment where the transplant took place, and have a life history that facilitates dispersal. As a result, the recolonization study provides clarity that early recolonization can occur, but does not provide further insight about the degree to which further succession is expected, nor the relative influence of chemical and physical stressors.

Ultimately, the benthic community information (both in situ and recolonization study samples) suggest that neither physical variables nor chemical constituents have presented an insurmountable obstacle to the early stages of recolonization. The benthic community structures in both Polley and Quesnel Lake are still in a state of flux where ecological factors (e.g., proximity of colonist sources; differences in dispersal strategies; competitive ability) appear to be influencing the overall progress in recolonization. Ultimately these factors may be as or more important than the current physical and chemical conditions in sediment. The case studies evaluated indicate that, under these conditions of response to a large scale disturbance, benthic community structure (even following succession) is unlikely to match the detailed compositional characteristics observed in regional reference areas. Instead, broad functional measures may provide a superior basis to evaluate overall ecological health, productivity, and successional stage during recovery.

[72] Ephermopteran (*Baetis rhodani*), plectopteran (*Nemoura cambric, Hydropsyche pellucidula*), isopod (*Asellus aquaticus*).

4.7.2.7 Risk Conclusions and Uncertainty Analysis

The initial scouring and deposition of tailings caused a significant loss of benthic organisms in all impacted waterbodies. Recovery is occurring but has not yet progressed to the point that impairments are absent. Risk conclusions for specific waterbodies include:

- **Hazeltine and Edney Creeks**—Edney Creek shows clear evidence of recolonization, and the density and Simpson's diversity index values in 2016 were not significantly different than the upstream reference location. Benthic recolonization of flowing creeks is generally rapid because upstream organisms drift downstream and provide a continued source of colonizing organisms. The Hazeltine Channel was only sampled in 2015 and is the subject of large-scale rehabilitation. However, recolonization of the Hazeltine Channel would also be expected to proceed relatively quickly.

- **Polley Lake**—Polley Lake also shows clear evidence of recolonization based on the density and Simpson's diversity index values in 2016 relative to Bootjack Lake. Recolonization of a lake environment is expected to be slower than in creeks because the absence of water flows and organism drift that would act as a source of active recolonization. Instead, different organisms will colonize the new habitat created by the deposition of tailings depending on their ability to disperse. The distance between the source of colonists and the new habitat will be a factor. Polley Lake would be expected to recolonize faster than Quesnel Lake West Basin because the total area of impact was smaller, and the average distance from the middle of the impacted area to the shoreline is lower. This expectation is consistent with the early indicators in the 2016 benthic data, which suggest an intermediate degree of recovery in Polley Lake (relative to the Hazeltine Channel and Quesnel Lake), and slower rates of recovery in the profundal zones of Polley Lake.

- **Quesnel Lake**—Quesnel Lake is also showing evidence of recolonization, but at a slower rate than Polley Lake. Three of the four areas have densities that continue to be at least one order of magnitude lower than their respective reference area. However, densities of organisms have increased from values near zero in 2014 to a value of several hundred organisms per square meter in 2016. The transplant study also showed that sediment from the exposed area would be colonized by organisms to a density of several thousand organisms per square meter if there is a nearby source of colonists.

As noted in this section, there was high variability in density and diversity amongst replicate stations within an area, as well as high variability within the reference areas amongst years. This variability is present despite the inclusion of several replicate samples for each sampling event, as well as considerable effort on the part of Minnow to match reference locations to the study area as closely as possible in terms of physical characteristics such as depth and grain size. The data were collected according to standard methods with appropriate quality assurance/quality control measures incorporated, and considered suitable for use for risk management decisions. A caveat is that effect determinations for the recolonization trials should not be based on categorical "pass-fail" interpretations of statistically significant differences or effect sizes. The data are not currently amenable to reliable situational evaluation of concentration-response; exploratory statistical analyses are also not recommended given that data has been collected from a limited number of locations as part of a pair-wise study design.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Benthic Community	Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings	Measure diversity and abundance in areas where tailings have been deposited (field-collected samples)	Diversity and abundance has not been reduced by more than 20% relative to background	Polley Lake	Abundance relative to background improved between 2014 and 2016, and was not reduced by more than 20% in the 2016 sampling. Diversity has also improved over time, but is still reduced by more than 20% in the three of the four sampling areas.
				Quesnel Lake	Abundance relative to background improved between 2014 and 2016, but was still reduced by more than 20% in the 2016 sampling in three of the four sampling areas. Diversity has also improved over time and was not reduced by more than 20% in the 2016 sampling in three of the four areas.
				Hazeltine Channel	Data were limited to sampling in 2015. Abundance was not reduced by more than 20% relative to Edney Creek background. Diversity was reduced by more than 20%.
				Edney Creek	Abundance and diversity were not reduced by more than 20% relative to Edney Creek background in the 2016 sampling.
		Measure diversity and abundance in areas where tailings have been deposited (transplant study)		Quesnel Lake	Transplant study demonstrated that deposited material with elevated tailings content may present a barrier for early stage recolonization by some benthic organisms.

4.7.3 Bioaccumulation in Benthic Organisms

Measurement of tissue chemistry for benthic invertebrates provides information on the bioavailability of sediment-associated metals, potential risks to benthic invertebrates, and chemical quality of food available to higher trophic level organisms. Metal concentrations in benthic invertebrates residing in waterbodies impacted by the breach were initially characterized in 2014 in the Quesnel River (MPMC 2015), followed by more extensive sampling in 2015 in Polley Lake, Quesnel Lake, the Hazeltine Channel, Quesnel River, and corresponding reference areas (Appendix N-1). Sampling in 2016 built upon previous work by characterizing metal concentrations in benthic invertebrates from Polley Lake, Quesnel Lake profundal near-field, the Hazeltine Channel, and corresponding reference areas (Appendix N-2).

4.7.3.1 Study Design and Methodology

The collection of benthic invertebrates for tissue analysis was conducted by Minnow in 2014, 2015, and 2016. Sampling locations and sample collection methodology are summarized below, with details provided in Appendix N-1 and N-2.

- In 2015, benthic invertebrate samples were collected from six depositional sampling areas: Polley Lake (POL-P2), Bootjack Lake (BOL-B2, reference), Quesnel Lake littoral, far-field (LFF, towards Likely) and profundal, near-field (PNF, near Hazeltine mouth) and corresponding reference areas (Horsefly Bay LREF1, PREF1)[73]. The same stations were sampled in 2016, except that the profundal, near-field station was moved further from the mouth of Hazeltine and the Quesnel Lake littoral areas were not sampled. To reduce variability among areas and stations, only chironomids (the dominant invertebrate) were retained for chemical analysis in 2016[74]. Sediments were collected with a stainless steel petite ponar or standard ponar and grab samples were placed in sieve bags (250- or 500-µm mesh size[75]) and sieved free of as much material as possible.
- In 2015, benthic invertebrate samples were collected from five erosional sampling areas: upper and lower Hazeltine Channel (HAC-U, HAC-D), lower Edney Creek (EDC, reference upstream of the area impacted by the breach), Quesnel River (QUR-1, exposed), and Cariboo River (CARU, reference). In 2016, only the Hazeltine and Edney creek stations were sampled. Samples were collected using combination of kick and sweep methods with a 400-µm mesh net and/or hand picking of benthic invertebrate from the underside of rocks. Based on dominant taxa present in each area in 2016, only caddisflies were retained for analysis in the Hazeltine Channel samples and only stoneflies were retained for Edney Creek samples[76].
- Samples were composited until the approximate tissue weight required for laboratory chemical analysis (0.2 g wet weight) was obtained. Samples were stored cool prior to further processing at the field laboratory and frozen prior to chemical analysis. In 2015 and 2016, biomass of benthic invertebrates at depositional stations was estimated on an area basis (g/m^2) using the surface area of sediment sampled as well as the sample weights.

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[73] See Figure 1 and Table A.1 of both Appendix N-1 and N-2 for details of sample locations

[74] Chironomids were the dominant taxa in depositional environments in both years, so this has little impact on data interpretation

[75] 250-µm mesh was used for deep samples and 500-µm mesh used for shallow samples to optimize retention of benthic invertebrates while minimizing retention of debris

[76] Previous work by Minnow in 2014 showed that metal concentrations in whole benthic communities from Quesnel River were similar to those in stoneflies for most metals (MPMC 2015)

- Stomach contents of invertebrates collected in 2014 and 2015 were not depurated prior to sample storage for analysis. In 2016, benthic samples were split into non-depurated and depurated organisms to evaluate the potential contribution of gut contents to metal concentrations in tissue. Organisms that were to be depurated were placed in site water with debris in a refrigerator until depuration could begin. Invertebrates were then picked from the debris and placed in site water for 24 h of depuration. Food was not provided and water was changed at 3 and 12 h to minimize re-ingestion of expelled gut material. Samples were then frozen for chemical analysis.
- Concurrent with tissue collection, sediment samples were collected from depositional areas and water samples collected from erosional areas for chemical analysis. Sediment and water chemistry samples were collected and analysed using the same methods as other routine monitoring programs, except as noted below.
- Tissue samples were freeze dried prior to analysis of metals by high-resolution inductively coupled plasma mass spectrometry (HR-ICP-MS). Analysis of 2014 samples were completed by the Saskatchewan Research Council Analytical Laboratories. Analysis of 2015 and 2016 samples were completed by the University of Missouri Research Reactor (MURR, Columbia, MO), which had the capacity to analyze small samples at low method detection limits. In 2016, sediment collected at each site was analysed by MURR using the same protocol as that used for tissue samples.
- Data analysis by Minnow focussed on selected parameters of interest (MPMC 2015). Principal Components Analysis was used as a supporting tool in 2014 and 2015 to determine if these parameters explained similarities and differences in benthic invertebrate tissue chemistry among stations. Tissue metal concentrations for most stations were statistically compared between exposed and reference areas. For the Hazeltine Channel, comparisons were made to available historical data (i.e., pre-breach collected in 2010). Tissue concentrations were related to exposure concentrations using the following ratios: biota-sediment accumulation factors (BSAFs) for depositional areas or bioconcentration factors (BCFs) for erosional areas[77]. In 2016, BCFs were also calculated for both depositional and erosional areas using DGT-labile concentrations measured at the sediment-water interface in 2015 and 2016 (referred to as DGT-BCFs; see Section 4.2.2 for discussion of DGT). For copper, the relationships between BSAFs or BCFs and exposure concentrations were examined.

4.7.3.2 Implications of Gut Purging on Tissue Concentrations

Although metal concentrations measured in non-depurated benthic invertebrates accurately represent dietary concentrations for higher trophic level organisms that consume benthos, it may not accurately represent metal concentrations in tissue of benthic organisms due to the potential contribution of gut contents and surface adhesion of metals and particulates. The potential for biological effects of metals in benthic invertebrates is typically attributed only to metal concentrations in body tissues, and not to the metal concentrations in sediment that may be present in the gut due to ingestion (Chapman 1985; 2016). The 2016 tissue collection by Minnow included depurated and non-depurated samples to evaluate the extent of contribution of gut contents to total metal concentrations in benthic invertebrates.

[77] BSAF = tissue concentration/sediment concentration; BCF = tissue concentration/water concentration; tissue and sediment concentrations on a dry weight basis; water concentration is dissolved metal

Depuration resulted in no significant differences in tissue concentrations of all analytes measured in benthic invertebrates collected from depositional environments (see Table 1 of Appendix N-2 [78]). In lotic samples (i.e., the Hazeltine Channel), depuration resulted in significantly lower concentrations of arsenic (11 and 37% lower for upper and lower creek samples) and copper (32% for lower creek). These decreases due to depuration were within the ranges previously observed for other metals and metalloids (e.g., Hare et al. 1989; Neumann et al. 1999). Overall, the depuration procedure resulted in only minor differences in benthic invertebrate tissue concentrations in depositional and erosional environments, with a few exceptions. Therefore, the focus of the remaining data analysis was on non-depurated tissue concentrations, which provides a conservative basis for evaluation.

4.7.3.3 Summary of Findings

Detailed results of tissue metal analysis and statistical analyses conducted by Minnow are provided in Appendix N-1 and N-2. Only tissue results for copper and arsenic are discussed below as these were identified as COPCs for sediment.

Polley Lake

Benthic invertebrate samples collected from Polley Lake and Bootjack Lake were composed mainly of chironomids (midge larvae) and oligochaetes in both collection years. In 2015, the approximate mean biomass (wet weight) of benthic invertebrates samples from the south end of Polley Lake (<0.17 g/m^2) was lower than reference samples from Bootjack Lake (>0.85 g/m^2). In 2016, there was a smaller difference in mean biomass of Polley Lake samples (>0.22 g/m^2) compared to those from Bootjack Lake (>0.38 g/m^2)[79].

Metal concentrations in benthic invertebrates from Polley Lake were not significantly different between 2016 and 2015, but were typically lower and less variable in 2016. In 2015, mean concentrations of copper (but not arsenic) in benthic invertebrates from Polley Lake were more variable and significantly higher than those from Bootjack Lake[80]. Principal Components Analysis of the 2015 data confirmed the distinction between these areas. In 2016, mean tissue concentrations of copper and arsenic were significantly higher for Polley Lake than Bootjack Lake[80]. However, arsenic concentrations in tissue samples from Polley Lake were lower in 2016 than 2015 and were similar to or less than Quesnel Lake reference samples. This suggests that the arsenic concentrations in benthic invertebrates in Polley Lake are within the natural range of conditions expected for the area surrounding the Mine, a finding that is consistent with other lines of evidence that indicate a lack of relationship between the tailings debris field and bioavailable arsenic contamination.

Both copper and arsenic concentrations were lower in benthic invertebrates than in sediments as indicated by BSAFs less than 1. BSAFs for the 2016 data were either similar to or lower than for the 2015 data. Mean copper concentrations of benthic invertebrates from Polley Lake were approximately 6-times higher than those from Bootjack Lake (Figure 55), consistent with the differences in sediment concentrations that are approximately 2-times higher in Polley Lake. The copper BSAF for Polley Lake was 4-times higher and DGT-based BCF was 2-times higher than for the reference, which suggests that copper in Polley Lake may be more available to benthic invertebrates compared to the reference. This is also consistent with higher concentrations of DGT-labile copper in water overlying the sediment (Section 4.2.2).

[78] Refer also to Tables D.1 to D.7 of Appendix N-2

[79] Refer to Table A.1 of both Appendix N-1 and N-2.

[80] Refer to Table 1 of Appendix N-1 and Table 2 of Appendix N-2.

Hazeltine Channel

Benthic invertebrate samples collected from upper Hazeltine Channel in 2015 were dominated by a high abundance of black fly larvae, whereas a more diverse benthic invertebrate assemblage, composed mainly of caddisflies and mayflies, was present in lower Hazeltine Channel. In 2016, caddisflies were the most abundant taxon in the Hazeltine Channel (based on biomass) and were selectively retained for analysis. Reference samples from Edney Creek were composed of a diverse mix of organisms, dominated by mayflies, stoneflies, and caddisflies, as well as chironomids in 2016. Stoneflies were the most abundant taxon in Edney Creek in 2016 (based on biomass) and were selectively retained for chemical analysis, due to low caddisfly abundance.

Metal concentrations in benthic invertebrates from both Hazeltine Channel and Edney Creek were typically higher in 2016 than 2015, and were significantly higher for some parameters. Mean concentrations of copper and arsenic were significantly higher in benthic invertebrates from the Hazeltine Channel than those from Edney Creek, and were generally similar in upper and lower Hazeltine Channel[81]. Principal Components Analysis of the 2015 data confirmed the distinction between the two exposed areas and reference area. However, tissue concentrations of most metals (including arsenic) were higher in historic (2010) samples from the Hazeltine Channel than samples collected from either creek in 2015 and 2016. Copper was the only metal for which mean tissue concentrations in 2015 and 2016 for both upper and lower Hazeltine Channel were significantly higher than pre-breach concentrations.

The higher copper concentrations in tissue are expected based on elevated copper concentrations in the Hazeltine Channel water. Mean concentrations of copper in the Hazeltine Channel tissue samples were approximately 6 to 12-times higher than Edney Creek and 5-times higher than historic samples. Copper BCFs were higher in the Hazeltine Channel than Edney Creek, but suggested little difference in the bioavailability of copper to benthic invertebrates in the Hazeltine Channel relative to historic conditions (Figure 4 of Appendix N-2). Examination of historical and post-breach data for all erosional areas (i.e., including Quesnel and Cariboo rivers) showed an inverse relationship between BCFs and aqueous concentrations and indicated that the Hazeltine Channel BCFs were only slightly higher than predicted from the relationship (Figure 4 of Appendix N-2). DGT-based BCFs (Figure 3 of Appendix N-2) for exposed and reference areas were consistent with those generally predicted by DGT-labile copper concentrations. Overall, the results suggested that copper was slightly more available to benthic invertebrates in the Hazeltine Channel relative to Edney Creek.

Quesnel Lake – Littoral

Benthic invertebrate samples collected from littoral areas of Quesnel Lake were composed mainly of chironomids, mayflies, leeches, amphipods, and pea clams. The approximate biomass of benthic invertebrate samples was similar between exposed and reference littoral areas in 2015[82]. Mean concentrations of metals in benthic invertebrates, including copper (Figure 55), did not significantly differ between samples from the littoral far-field area (LFF) and Horsefly Bay reference area (LREF1)[83]. Principal Components Analysis supported these results with little distinction between the exposed and reference areas. BSAFs were also similar between areas. Sediment concentrations of copper in the littoral exposed area were typically below the CSR sensitive sediment criterion (see Section 4.6.1).

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[81] Refer to Table 2 of Appendix N-1 and Table 3 of Appendix N-2.

[82] Refer to Table A.1 of Appendix N-1 and N-2.

[83] Refer to Table 1 of Appendix N-1 and Table 2 of Appendix N-2.

Quesnel Lake – Profundal

Benthic invertebrate samples collected from profundal areas of Quesnel Lake were composed mainly of chironomids. In 2015, the approximate mean biomass (wet weight) of benthic invertebrates samples from the profundal near-field area (<0.09 g/m^2) was lower than the profundal reference area in Horsefly Bay (>0.34 g/m^2; Appendix N-1). In 2016, there was a smaller difference in mean biomass of exposed samples (>0.34 g/m^2) compared to those from the reference area (>0.70 g/m^2)[82].

Metal concentrations in benthic invertebrates from the profundal near-field areas were not significantly different between 2016 and 2015, despite the change in sampling location, but were typically lower and less variable in 2016. Copper was the only metal for which mean concentrations in benthic invertebrates from the profundal near-field areas (PNF) were significantly higher than those from the reference area (PREF1)[83].

Mean copper concentrations of benthic invertebrate from the profundal near-field areas were approximately 3-times higher than those from the profundal reference area (Figure 55), despite the 16- to 21-fold difference in sediment concentrations of copper between these areas, based on sediment samples collected concurrently with tissue. Copper BSAFs were less than 1 and lower for the near-field area than reference area, consistent with the expected inverse relationship between BSAF and sediment concentrations (Figure 2 of Appendix N-2). The DGT-BCF for the near-field area was 1.7-times higher than for the reference which suggests the bioavailability of copper in overlying water is slightly higher in the near-field area (Appendix N-2).

Quesnel River

Benthic invertebrate samples collected from Quesnel River and the corresponding reference area in Cariboo River were more diverse than samples collected from the Hazeltine Channel, and were composed mainly of stoneflies, mayflies, caddisflies, and snails.

In 2014, copper was the only metal for which the mean concentration in benthic invertebrates from Quesnel River (QUR1) was significantly higher than the reference area (CARU). However, the difference in tissue concentrations was less than a factor of 2. Tissue metal concentrations in 2015 were slightly lower, but were not significantly different from 2014. In 2015, mean tissue concentrations of metals, including copper (Figure 55), did not significantly differ between the Quesnel River and the reference area[84]. Principal Components Analysis of the 2015 data indicated little to no distinction between areas.

[84] Refer to Table 2 of Appendix N-1.

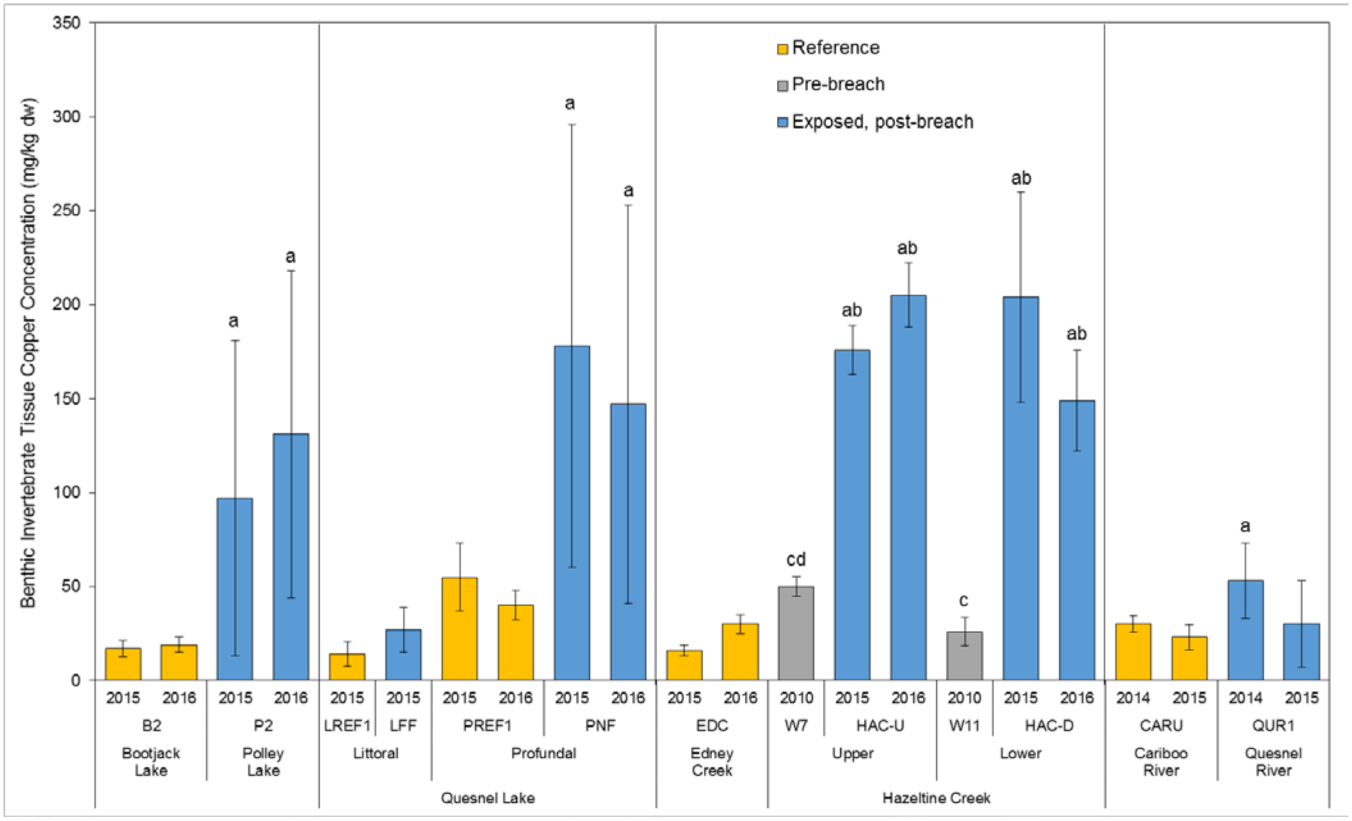

Figure 55: Tissue Concentrations of Copper in Benthic Invertebrates from Polley Lake, Quesnel Lake, the Hazeltine Channel, and Quesnel River, 2014 to 2016

Note: Adapted from Figures 2 and 4 of Appendix N-1. Data are mean dry weight concentrations and *t**standard error (i.e., width of 95% confidence interval). NF = near-field; FF = far-field; a = significantly different from corresponding reference for same year; b = significantly different from pre-breach; c,d = significantly different from corresponding reference, for 2015 and 2016 respectively.

4.7.3.4 Relative Contribution of Physical and Chemical Stressors

This particular line of evidence is focused on chemical stressors only. Physical stressors are not a factor that could confound the interpretation of these data as the benthic invertebrate tissue chemistry reflect the relative uptake and bioavailability of sediment-associated metals. Physical stressors may potentially influence the abundance and composition of the benthic community (reported in Section 4.7.2) that would be present for tissue collection.

4.7.3.5 Uncertainty Analysis

Other lines of evidence provide indirect estimates about how much metals are likely to accumulate in the food chain from sediment; however, measurement of tissue concentrations provides a more direct approach to assessing potential risks to benthic invertebrates. Metal concentrations can be compared between the management and reference areas to evaluate the relative increased in exposure of benthic invertebrates.

Benthic invertebrate samples were collected following well-established field methods, using consistent approaches across sampling events. Metal concentrations in tissues were measured using standardized analytical test methods with the capacity to analyze small samples at low method detection limits and robust QA/QC practices. Minnow completed a data quality assessment upon receipt of chemistry results and determined that results met data quality objectives and was considered to be acceptable for the use of interpretation and derivation of conclusions, but noted some heterogeneity and high natural variability in the 2015 data.

In 2016, dominant taxa were specifically selected from samples for chemical analysis in an effort to minimize variability in tissue concentrations. This difference in sample processing between years was considered to have little impact on data interpretation because in depositional areas the same dominant taxon was present both years and the split analyses in erosional areas in 2014 suggested little difference in tissue concentrations of whole benthic communities versus a specific taxon.

Benthic invertebrates that ingest sediment to extract organic carbon likely reflect sediment metal concentrations in their gut, rather than only the metals that have been incorporated into tissue and are available for toxicity. The 2016 tissue analysis included depurated and non-depurated samples and demonstrated that depuration made little difference in benthic invertebrate tissue concentrations in the context of this study.

Changes in tissue chemistry does not necessarily mean that an adverse effect is occurring at the individual or population level (which were evaluation in Sections 4.7.1 and 4.7.2). Therefore, the strength of association to the assessment endpoint is considered low to moderate.

4.7.3.6 Comparison to Decision Criteria

The decision criterion for the benthic invertebrate chemistry line of evidence focused on statistical differences between exposure and reference areas.

Regarding concentrations of copper in benthic invertebrates (Figure 55), mean tissue concentrations in samples from Polley Lake, Quesnel Lake profundal areas, and the Hazeltine Channel were significantly different and more than 20% higher than mean concentrations in samples from the corresponding reference areas. In the littoral areas of Quesnel Lake, the mean concentration in the far-field sample was not significantly different from the reference sample despite being approximately 2-times higher. The mean concentrations in Quesnel River samples were significantly different from the Cariboo River references samples only in 2014. In both 2014 and 2015, the difference in mean concentrations compared to reference samples was less than 2 fold; therefore, these were not considered to be meaningfully different.

As described in Section 4.7.3.6, the decision criterion was set based on no statistical difference or no more than a 20% higher tissue concentration relative to reference samples. Copper was the only metal where differences in concentrations between exposed and reference areas were observed.

Line of Evidence	Assessment Endpoint	Measurement Endpoint(s)	Decision Criterion	Area	Decision
Benthic invertebrate tissue chemistry	Protect benthic invertebrates from reductions in survival, growth and reproduction as a result of direct contact with tailings.	Measure the concentration of metals in field-collected benthic invertebrates.	The average concentration in samples collected from the management area is not statistically different OR not more than 20% higher than the average concentration in samples collected from reference areas.	Polley Lake	>20% higher and significantly different for copper
				Hazeltine Channel	>20% higher and significantly different for copper
				Quesnel Lake littoral	Not statistically different
				Quesnel Lake profundal	>20% higher and significantly different for copper
				Quesnel River	Not statistically or meaningfully different

4.8 Effects-Based LOE: Higher Trophic Level Aquatic Receptors

4.8.1 Food Chain Modelling

Several piscivore wildlife species were evaluated as part of the mechanistic food chain model described in Section 3.4.2. Golder has opted to not repeat duplicate information in this section, but has highlighted any aquatic-specific differences. As with the terrestrial wildlife receptors, a detailed discussion of the food chain model is provided in Appendix J.

There were no differences in study design or methodology relative to the discussion provided in Section 3.4.2.1. The wildlife receptors are summarized in Table 46. There were no differences with respect to the decisions with respect to receptor parameterization or the selection of toxicity reference values. Reasonable worst-case exposure concentrations are described in detail in Appendix J, but in brief:

- The assumed drinking water concentrations are the 95% UCLM concentration from the Hazeltine Channel. This is the same assumption as described for the terrestrial wildlife, but is more conservative because piscivores are likely obtaining their drinking water from a wider range of aquatic environments.
- Dietary concentrations are based on site-specific data. The available fish tissue data was divided into "small-bodied fish" and "large-bodied fish" and the 95% UCLM was calculated for the totality of "impacted" and "background" locations.
- Wildlife were assumed to have negligible incidental ingestion of sediment
- The relative bioavailability of the biologically-incorporated metals was assumed to be the same as presented for terrestrial receptors.

Table 46: Wildlife Receptors Retained for Food Chain Modelling

Receptor Group	Feeding Guild	Surrogate Receptor
Birds	Piscivorous	Osprey, Great blue heron
Mammals	Piscivorous	River otter

Table 47: Summary of Hazard Quotients for Aquatic Wildlife - Copper

Species	Exposed Aquatic Habitats	Background Aquatic Habitats
Osprey	0.01	0.01
Great blue heron	0.01	0.01
River otter	0.01	<0.01

The results of the food chain model for piscivores are subject to the same challenges and sources of uncertainty as described for the terrestrial wildlife receptors. A check was conducted to determine if the risk conclusion would change if piscivores were consuming amphibians instead of fish, and no change in the conclusion was noted. Golder concludes that risks to wildlife associated with ingestion of water and fish are unlikely to be present based on the hazard quotients summarized above. The food chain model was constructed with reasonable worst-case exposure concentrations and appropriately conservative assumptions. Hazard quotients were substantially less than one for all receptor/contaminant of potential concern combinations and were also indistinguishable from background.

4.9 Weight-of-Evidence Integration

4.9.1 Weight of Evidence Process

The following sections provide a summary of the weight and magnitude of response for each line of evidence as applied to each major ecosystem component. The weight of evidence approach is based on guiding principles described by provincial guidance supporting Protocol 20 (SABCS 2008, 2011) and was described by Golder during the engagement with technical reviewers (Appendix F-2). In brief, the weight of evidence process started with assigning a rating of “high”, “moderate”, or “low” to each line of evidence based on weight, magnitude and overall uncertainty.

- A preliminary weight was assigned in the problem formulation by considering: a) how closely the line of evidence matched the assessment endpoint, and b) the likely uncertainty associated with the line of evidence based on sensitivity and specificity, study design and data quality objectives, representativeness, and correlation and causation. These preliminary weights were re-evaluated at the conclusion of the exposure and effects assessment based on the overall quantity and quality of information obtained.

- The magnitude of response for each line of evidence by comparing the data to a decision criterion that specified when the magnitude of the observed effect would be considered a potential risk. Some decision criteria are specified by provincial guidance, while others required consideration of the available literature to determine a criterion that was ecologically relevant. Decision criteria can be quantitative or qualitative, depending on the circumstances of each line of evidence.

- The uncertainty in the line of evidence was based on the degree to which it was used to inform the risk assessment conclusion. Broadly, a line of evidence was considered more certain if the data were sufficient to extrapolate from specific sampling locations to a broader spatial scale, if it was able to clarify the relative influence of physical versus contaminant stressors, and if it there was a quantitative basis for evaluating the magnitude of response against a regulatory-approved decision criterion.

Weight-of-evidence assessment does not need to be a quantitative, prescriptive process. Best professional judgement is part of the process. To provide transparency, each individual line of evidence was accompanied by a narrative that describes the site-specific data and relevant information from the peer-reviewed literature. A key question in the narrative is the degree to which the available data supports a causal relationship. Most decision frameworks incorporate Hill's (1965) criteria for causation in one form or another. Environment Canada (2013) provide guidance on integrating causality into ecological risk assessments for contaminated sites. Establishing causality is itself a weight of evidence based on observational data, experimental results from manipulation, or alignment with general knowledge from the scientific literature (Environment Canada 2013b). Forbes and Calow (2004) conclude that causality is present if there is a correlation in field data that can be confirmed through a controlled experiment and that is consistent with the known scientific information. If the evidence for causality is weak, it would be difficult to establish a meaningful remedial strategy where actions related to specific contaminant or physical stressor would be likely to result in biological improvement.

4.9.2 Consideration of Physical Stressors and Contaminant Bioavailability in the Risk Assessment

The scouring of Hazeltine Creek and the lower portion of Edney Creek, and the deposition of tailings and native soils into the lake and creek waterbodies resulted in significant physical impacts to all components of the aquatic ecosystem (microbes, fungi, invertebrates and plants) within the spatial bounds of the affected area. Golder concluded that is not necessary to quantify the magnitude of this impact (beyond the qualitative observation that it was substantive and wide-spread) as part of this ecological risk assessment. Further information about the total area of impacted aquatic environment can be found in MPMC (2015). The aquatic environments of Hazeltine Creek has also been the focus of an extensive rehabilitation program.

The purpose of the risk assessment was to determine if the contaminant stressor (i.e., residual concentrations of metals in tailings) presented an unacceptable risk that requires risk management over and above the rehabilitation and monitored recovery program that is already underway to address the physical stressors (nutrient and physical deficiencies) that are inherent from the scouring and deposition event. The aquatic ecosystem will gradually recolonize the impacted areas once the environment has been reconnected. In the case of Polley and Quesnel Lakes, fish are mobile and will start to immediately use any accessible habitat. Zooplankton and phytoplankton communities would also expand to occupy those areas. Sediment recolonization occurs at a slower rate because sediment organisms are less mobile, and colonization may be limited by the physical deficiencies of the substrates. In Edney and Hazeltine Creek, reestablishment of the benthic community may be faster than in the lakes because aquatic organisms can drift downstream from established source populations. Fish will also tend to reoccupy areas quickly once barriers to their passage have been removed. An unacceptable risk, in this context, was related to whether the effects from chemical contamination or physical stressors will prevent the long-term successful re-introduction of a biologically diverse, functional, self-sustaining and inter-dependent aquatic ecosystem.

4.9.3 Primary Producers and Zooplankton

A summary of the lines of evidence is provided in Table 48. Overall, the weight of evidence indicates that residual risk to primary producers and zooplankton from COPCs is relatively low. Water concentrations of COPCs are lower than WQGs and therefore primary producers and zooplankton are not exposed to hazardous conditions from direct contact with water (Assessment Endpoint 1). Toxicity testing indicates that direct acute and chronic effects did not occur under laboratory conditions, and abundance and biomass of field-collected samples suggest that the reductions in survival, growth and reproduction are not occurring (Assessment Endpoint 3).

Table 48: Summary of Lines of Evidence for Primary Producers and Zooplankton

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there concentrations of metals that are hazardous in water? (Section 4.2.1)	Quantitative—Comparison to ambient WQGs	Low—Although water chemistry data are indicative of exposure conditions, concentration data do not provide direct information on potential effects.	Low for Polley Lake, Quesnel Lake and Quesnel River —the 95th percentile concentration of copper was lower than WQGs. Moderate for Hazeltine Channel – the 95th percentile concentration of copper was greater than the WQG.	Moderate—The total concentrations of metals in water is not particularly informative for the overall risk conclusion beyond providing a measure of potential exposure. However, there has been a substantial amount of sampling conducted which provides confidence that water chemistry is well understood.
Are those concentrations present in a form that could be taken up by plankton? (Sections 4.2.1 and 4.2.2)	No specific decision criterion. Supporting information for interpreting other LOE.	Moderate—Linkage to suspended solids and speciation modelling provide refined estimate of potential bioavailability of COPCs.	Low—Copper not found to be in form that can be readily taken up by biota.	High—There were multiple chemical surrogates that showed that copper and other metals were unlikely to be highly bioavailable to plants or zooplankton. The site-specific data was consistent with the underlying geochemical understanding of the material.
Are metals actually being accumulated by plankton? (Section 4.5.2)	Semi-quantitative. Comparison of Cu concentrations in field-collected samples to range from reference samples and to dietary threshold for fish. Statistical comparisons were not appropriate.	Moderate—Tissue chemistry are indicative of exposure (uptake), but concentrations do not provide information on potential effects.	Low—Tissue concentrations were within normal range and were lower than the dietary threshold for fish.	Moderate—There is high inherent variability in plankton communities due natural factors (e.g., temperature daylight) and due to natural patchiness.
Does water exert adverse effects to primary producers or zooplankton under laboratory conditions?	Quantitative—50% effect level for acute tests; 25% effect level for chronic tests	Moderate—Toxicity testing allows for the direct measurement of potential effects from water quality to the survival, growth, and reproduction of test organisms. However, toxicity testing is conducted under laboratory conditions and with cultured test organisms.	Low—Lower survival and reproduction in some test waters was observed in chronic tests but this was related to turbid conditions that no longer exist.	Moderate. Toxicity tests are a common line of evidence in ecological risk assessments, and there was considerable amounts of water toxicity data collected over space and time during worst-case exposure conditions.

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Is there evidence of adverse effects to the primary producers or zooplankton under field conditions?	Semi-quantitative—Biomass or abundance has not changed relative to background areas, based on visual examination of plots.	High—Provides a direct community-level of measurement of potential changes in plankton.	Low—No change in biomass or abundance observed.	Moderate—There is high inherent variability in plankton communities due natural factors (e.g., temperature daylight) and due to natural patchiness.

4.9.4 Benthic Invertebrates

A summary of the lines of evidence is provided in Table 49. Overall, the weight of evidence indicates that the magnitude of residual risk to benthic invertebrates ranges from moderate to high depending on the location of the community. The magnitude of impairment is largely a function of the degree to which the physical debris field affected the biologically active layer of sediment; this ranged from complete burial of the communities to physical modification of the substrate type where thinner deposits of debris accumulated. The largest responses were observed in the Hazeltine Channel and profundal Quesnel Lake where the greatest physical impacts occurred; however, all portions of the receiving environment have already exhibited signs of early succession (including within the reconstructed habitat of the Hazeltine Channel). Note that the designation of "moderate to high" risk is based on the current conditions measured by the post-breach monitoring and is not a prediction of the long-term risks once the environment returns to a more natural condition (e.g., deposition of organic carbon through lake productivity and suspended sediment transport).

Although there were obvious initial impacts from the tailings debris flow, the more salient question from a risk management perspective concerns whether the residual risk is primarily related to physical stressors or whether contaminant influences (particularly copper) may be exacerbating this risk (or provide a constraint to the restoration and succession of the natural communities in the long term) because the answer to this question will inform the remediation plan. To evaluate this question, a tiered (but overlapping) set of studies was conducted to evaluate chemical risk potential:

- Bulk Exposure (Assessment Endpoint 1a)—Bulk sediment concentrations of copper exceeded the $SedQQ_{TS}$ standards above large areas in Polley Lake, Quesnel Lake (littoral and profundal), and Quesnel River; therefore there was a potential for adverse effects via direct contact with sediment that required further evaluation with site-specific tools. Concentrations of remaining metals (e.g., arsenic) exhibited low potential for risk and exhibited weak to non-existent spatial relationship to the TSF embankment breach; therefore the assessment of stressors was easily narrowed to copper and physical effects.
- Chemical Availability (Assessment Endpoint 1b)—The magnitude of exceedance of generic screening criteria was not large in most sediments, indicating that constraints to leachability or bioavailability of copper in sediment may be effective in ameliorating the chemical hazard. Surrogate measurements of how much of the total concentration is likely be mobile or bioavailable (e.g., leaching potential from tailings; sequential extractions; bioavailability models such as AVS-SEM) indicated that only a small portion of the sediment-associated copper is available for uptake.

- Effects-Based Measures (Assessment Endpoint 2)—Toxicological effects in standardized laboratory toxicity tests have been observed, although the confounding factor of substrate type has limited the ability to discriminate chemical factors from grain size and TOC influences. Studies of benthic community health and recolonization status have also been conducted; these indicate that tailings-influenced sediments are improving over time, with the greatest rate of improvement observed in areas that have either been rehabilitated (the Hazeltine Channel), have proximity to unimpacted areas to support recolonization, and/or that had lower degree of disruption from the tailings debris flow.

Copper appears to be contributing to effects on the benthic community in the low organic carbon profundal areas of Quesnel Lake; however, the relative influence of the physical factors such as low organic carbon is substantially greater than copper. Copper is not expected to be a limiting factor for the natural recovery of the benthic community, based on the information collected to date.

Table 49: Summary of Lines of Evidence for Benthic Invertebrates

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there bulk concentrations of metals that are hazardous in sediment?	Quantitative—Comparison of bulk chemistry to Schedule 9 CSR numerical standards and to reference conditions.	Low—The concentrations of total copper do not provide direct information on potential effects, particularly given the elevated reference copper concentrations and concentrations within an order of magnitude of the $SedQC_{SS}$.	Moderate—Although large areas of aquatic sediment exceed $SedQC_{SS}$ and $SedQC_{TS}$ for copper, the magnitude of exceedance was seldom large, with most contamination in the Quesnel West Basin <5x $SedQC_{SS}$. Concentrations of other constituents (e.g., arsenic) are not present at level of concern.	Low—Bulk sediment chemistry was considered for screening of COPCs are areas of greatest hazard, but bioavailability and effects-based tools are more reliable.
Are those bulk concentrations present in a form that could be taken up by benthic invertebrates	Quantitative—Comparison of porewater chemistry to ambient WQGs. Measures of bioavailability from sequential Tessier extractions, SEM:AVS.	Moderate—The measures of Cu leachability, binding to sulphides, and uptake into aquatic biota provide an indication of whether and to what degree Cu is mobile or sequestered. However, concentration data do not provide direct information on potential effects.	Moderate—Cu in porewater greater than 20% different between exposed and reference locations, but the magnitude of difference was less than 3-times. Sequential extractions indicate low potential for Cu to dissociate into porewater. Sulphides expected to bind Cu in Polley Lake and Quesnel Lake littoral sediments. SEM-AVS not applicable in oxidized profundal sediments of Quesnel Lake or Hazeltine Channel sedimentation pond.	Moderate—There were multiple chemical surrogates that showed that copper and other metals had low bioavailability to benthic invertebrates. The site-specific data were consistent with the underlying geochemical understanding of the material and with the patterns of bioaccumulation observed in aquatic tissues.
Are metals actually being accumulated by benthic invertebrates?	Quantitative—Relative and statistical comparison of Cu concentrations in field-collected samples to range from reference samples.	Moderate—Tissue chemistry are indicative of exposure (uptake), but concentrations do not necessarily provide information on potential effects.	Moderate—Mean tissue concentrations in samples from Polley Lake, Quesnel Lake profundal areas, and Hazeltine Channel were significantly different and more than 20% higher than mean concentrations in samples from the corresponding reference areas.	Moderate—The bioaccumulation data indicate that a portion of the Cu found in sediment is available for uptake into organisms (i.e. a low bioavailability is not zero). There is uncertainty with respect to whether increased tissue burdens (by approximately 5-times) would impair health or function of invertebrates.
Are there physical characteristics that could limit benthic invertebrates in the absence of elevated metals?	Semi-quantitative—evaluation of particle size distribution, TOC, and burial depth from TSF breach.	High—the short term response of benthic communities to large scale tailings deposits ranged from complete burial to modified physical habitat. This linkage will remain important as recolonization and restoration of habitat occurs.	High—In littoral areas of Quesnel Lake, particles size differences and very low TOC content (<1%) compared to far-field and reference areas are confounding factors for the interpretation of biological effects. TOC content in several areas (e.g., Quesnel Lake profundal, Hazeltine Channel) was very low (<1%).	High—Consideration of the modified substrate is important for both the identification of direct impacts related to burial and habitat modification, and also as a confounding factor for assessment of potential copper effects.

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Do sediments exert adverse effects to benthic invertebrates under laboratory conditions?	Quantitative—20% effect level compared to reference, or to laboratory controls when no appropriate reference sample.	Moderate—Toxicity testing allows for the direct measurement of potential effects from water quality to the survival, growth, and reproduction of test organisms. However, toxicity testing is conducted under laboratory conditions and with cultured test organisms.	Low—no significant toxicity in Polley Lake sediments. Moderate—potential risk in the Hazeltine Channel and Quesnel Lake (littoral and profundal), likely influenced by physical factors.	Moderate—Overall, the analyses to date show that observed toxicity responses are not solely related to chemical conditions and that the influence of physical factors (e.g., TOC) and chemical exposures (e.g., copper) are interrelated.
Is there evidence of adverse effects to the benthic invertebrates under field conditions?	Semi-quantitative—20% effect level in field measures of benthic health compared to reference. Results of recolonization studies are less quantitative.	Moderate—measure provide direct indicators of benthic community status. Main constraint is variability and confounding factors, and fact that communities are currently transitional during early successional stages.	Moderate to High—The initial scouring and deposition of tailings caused a significant loss of benthic organisms in all impacted waterbodies. Monitoring indicates statistically significant reductions (greater than 20% reductions) in both density and diversity in multiple water bodies. Recovery is occurring, with slowest response in profundal Quesnel Lake. Neither physical variables nor chemical constituents have prevented early stages of recolonization.	Moderate to High—The benthic community structures in both Polley and Quesnel Lake are still in a state of flux where ecological factors (e.g., proximity of colonist sources; differences in dispersal strategies; competitive ability) appear to be influencing the overall progress in recolonization.

4.9.5 Fish

A summary of the lines of evidence is provided in Table 50. Overall, the weight of evidence indicates that residual risk to fish is low. Water concentrations of copper are lower than WQGs and therefore fish are not exposed to hazardous conditions from direct contact with water in the lakes and streams assessed (Assessment Endpoint 1). Toxicity testing conducted under representative laboratory exposures relevant to current chronic conditions indicates that direct acute and chronic effects did not occur, and tissue concentrations of prey items and fish themselves do not indicate harmful levels of bioaccumulation in fish from either Polley or Quesnel Lakes (Assessment Endpoint 4). Histopathology and body measurements of field-collected samples indicate that the reductions in survival, growth and reproduction are not occurring in Quesnel Lake (Assessment Endpoint 4).

Although residual risk from exposure to contaminants is considered to be low, it is important to acknowledge that the TSF breach affected fish access to habitat as well as the physical habitat itself, in particular in Hazeltine Creek. The Hazeltine Creek corridor has been undergoing progressive reconstruction and at this time, access to fish habitat in undisturbed upper reaches of Edney Creek has been re-established and fish, including kokanee, coho and chinook salmon have been observed in the reconstructed portion of the channel as well as upstream reference areas. Hazeltine Creek channel reconstruction continues in the upper reaches and the access from Polley Lake to rainbow trout spawning habitat in upper Hazeltine Creek is still restricted.

Table 50: Summary of Lines of Evidence for Fish

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Are there concentrations of metals that are hazardous in water?	Quantitative— Comparison to ambient WQGs	Low—Although water chemistry data are indicative of exposure conditions, concentration data do not provide direct information on potential effects.	Low for Polley Lake, Quesnel Lake and Quesnel River —the 95th percentile concentration of copper was lower than the WQG. Moderate for Hazeltine Channel – the 95th percentile concentration of copper was greater than the WQG.	Moderate—The total concentrations of metals in water is not particularly informative for the overall risk conclusion beyond providing a measure of potential exposure. However, there has been a substantial amount of sampling conducted which provides confidence that water chemistry is well understood.
Are those concentrations present in a form that could be taken up by fish?	No specific decision criterion. Supporting information for interpreting other LOE.	Moderate—Linkage to suspended solids and speciation modelling provide refined estimate of potential bioavailability of COPCs.	Low—Copper not found to be in form that can be readily taken up by biota.	High—There were multiple chemical surrogates that showed that copper and other metals have low bioavailability
Are metals actually being accumulated by fish?	Semi-quantitative—Comparison of metals concentrations in field-collected samples to range from reference samples. Comparison to literature-based tissue toxicity thresholds.	Moderate—Tissue chemistry are indicative of exposure (uptake), but concentrations do not necessarily provide information on potential effects.	Low—Copper and arsenic in fish tissues found to be moderately higher in exposed areas than reference; however, concentrations were lower than TRVs.	Moderate.
Does water exert adverse effects to fish under laboratory conditions?	Quantitative—50% effect level in acute tests. 25% effect level in chronic tests.	Moderate—Toxicity testing allows for the direct measurement of potential effects from water quality to the survival, growth, and reproduction of test organisms. However, toxicity testing is conducted under laboratory conditions and with cultured test organisms.	Low—Water was not acutely lethal to larval rainbow trout. Water did not cause chronic effects to larval rainbow trout or fathead minnows.	Moderate—Repeated confirmation of non-toxic water using a sensitive chronic species provides confirmation of favorable water quality conditions.
Are there physical characteristics that could limit fish in the absence of elevated metals?	No specific decision criterion. Supporting information for interpreting other LOE.	Moderate—factors that influence the abundance or diversity of prey, or access to quality habitat are important.	Low to Moderate— Although productivity of benthic organisms as a food source is limited in some areas, communities are recovering and fish can forage over other areas during succession.	Low—For Polley Lake, the biggest influence on fish is likely the loss of access to upper Hazeltine Channel. Physical changes also occurred at Quesnel Lake as a result of the breach; however, access to fish habitat in Edney Creek has been restored.

Line of Evidence	Decision Criterion	Linkage to Assessment Endpoint	Magnitude of Response (Hazard or Effect)	Contribution to Risk Conclusion
Is there evidence of adverse effects to the fish under field conditions?	Qualitative comparison of incidence of histological abnormalities	Moderate—Histological condition can be a measure of disease which could be influenced by prolonged exposure to stressors (contaminants and particulates) in the environment.	Low—No difference between fish from exposed and reference locations in Quesnel Lake in feeding activity, tissue trauma, or presence of parasites.	Moderate—Significance uncertain as evidence of histological changes can have multiple causes including sampling artifacts (e.g., fish handling) or natural stressors (e.g., background incidence of parasites).
	Semi-quantitative comparison of condition factor to reference.	Moderate. Body metrics are measures of growth and energy storage which could be influenced by prolonged exposure to stressors in the environment. However, fish growth and foraging behaviour can be influenced by a variety of factors beyond contaminant exposure, including interspecies competition and variations in population density.	Low—No difference between fish from exposed and reference locations in either Polley or Quesnel lakes.	Moderate—Condition factor provides an overall integration of the response of a fish to positive (nutrition) and negative (chemical and physical stressors) influences. The relatively large sample size for sockeye salmon juveniles provides a good indication that fish in the exposed area do not have a lower condition factor than those in reference areas and therefore are not affected by stressors. The sample size is relatively low for other fish species, but the results are similar.
	Semi-quantitative comparison of age-class to reference.	Moderate—Age-class provides a measure of potential reproductive effects based on fish health and habitat quality.	Moderate—The age frequency distribution for rainbow trout in 2016 was shifted to the right compared to the distribution for 2014, with an apparent absence of rainbow trout in age groups of 1 and 2 years old in 2016. This indicates a likely reduction in recruitment in 2014 and 2015, following the TSF embankment breach.	High.

4.9.6 Piscivorous Wildlife

There are low risks to piscivorous mammals and birds as result of copper accumulating in the aquatic food chain. This risk conclusion is based on the findings from a mechanistic food chain model that was constructed using default parameters established by regulatory guidance, along with site-specific information about metal concentrations in water and dietary items. The mechanistic food chain model achieved a defensible balance between using site-specific data to improve ecological realism while retaining conservative assumptions that reduce the chance of making an incorrect conclusion. There is no weight of evidence for evaluating risks to wildlife—numerical standards for water or sediment are not designed to directly evaluate this pathway. There was no direct measurement of tissue concentrations or experimental validation of effects under field conditions given the obvious ethical and logistical limitations of these approaches. Other lines of evidence with respect to risks to mammals and birds is not considered necessary in light of the low magnitude of hazards calculated by the food chain model.

4.9.7 Implications of Spatial Scale of Risk Conclusions

The preceding section provided a summary of the risk conclusions based on the magnitude of observed response relative to the decision criteria for individual lines of evidence. The evaluation focused on describing how different lines of evidence were contributing to the overall risk to the ecosystem on a site-wide basis. An uncertainty analysis was conducted for each individual line of evidence (and reported in preceding sections). The overall uncertainty in the line of evidence reflected the degree to which it was used to inform the risk assessment conclusion. This type of uncertainty analysis is routine in an ecological risk assessment. However, in addition to the uncertainty associated with individual lines of evidence, there is an additional consideration for risk management in how risk conclusions based on the site as a whole can be used to inform site management planning at a less than site-wide basis.

Subareas, in this context, are less static than those considered in the terrestrial ERA—soil contamination is immobile, and can be readily divided into management units. Conversely, contamination in sediment or surface water can be mobilized and redistributed (in the case of sediment), or is inherently variable (in the case of surface water). Subareas in the aquatic context are likely focused on looking at specific portions of Hazeltine Creek in terms of water-borne exposure, or at different areas in Quesnel Lake in terms of sediment exposure. Each line of evidence varies in terms of its ability to support interpolation to a sub-area level to help support informed risk management decisions.

Table 51: Spatial Scale of Different Lines of Evidence

Line of Evidence	Decision Criterion	Spatial Coverage	Ability to Interpolate to a Subarea
Are there concentrations of metals that are hazardous in water?	Quantitative— Comparison to ambient WQGs	Moderate. Water sampling focused on temporal coverage at a relatively small number of representative locations.	High for lakes because water quality reflects a relatively static and large volume of water. Moderate for creeks because concentrations can vary depending on the specific locations of seepage zones and groundwater discharge areas. The total volume of water associated with the creek exposure scenario is also far less than in a lake.
Are those concentrations present in a form that could be taken up by plankton, invertebrates and fish?	No specific decision criterion. Supporting information for interpreting other LOE.	Moderate. Water sampling included both total and dissolved metals (see above). DGT samplers were deployed primarily in Polley and Quensel Lake with a limited number of samplers in Hazeltine Channel. Metal speciation modelling was limited to the lakes.	High for lakes because water quality is based on a large, static volume. Low for creeks because DGT samplers may not have focused specifically on areas where geochemical considerations would result in maximum bioavailability.
Are metals actually being accumulated by aquatic organisms?	Semi-quantitative. Comparison of Cu concentrations in field-collected samples to range from reference samples and to dietary threshold for fish. Statistical comparisons were not appropriate.	Moderate for plankton and benthic invertebrates in Polley and Quesel Lake. High for fish in Polley and Quensel Lakes because of the mobility of the organisms. No data collected for Hazeltine Channel.	High for the lakes. There was adequate spatial coverage in the lakes and mobile organisms integrate exposure over time and space.
Does water exert adverse effects under laboratory conditions?	Quantitative	Moderate. Toxicity testing was done in conjunction with water sampling, which focused on temporal coverage at a relatively small number of representative locations. No toxicity testing conducted on post-rehabilitation conditions in Hazeltine Channel.	High for lakes because water quality reflects a relatively static and large volume of water. Data can also be extrapolated based on water chemistry (i.e., if copper concentrations for a new set of samples are less than concentration included in the toxicity testing program, then it is reasonable to conclude that toxicity will also be low)
Does sediment exert adverse effects under laboratory conditions	Quantitative – 20% reduction relative to negative controls	Moderate. Sediment toxicity testing focused on achieving replication for representative areas, not spatial coverage of the entirety of the different lakes.	High. Toxicity testing focused on developing the concentration response relationship between the stressors (copper; low TOC) and effects.

Line of Evidence	Decision Criterion	Spatial Coverage	Ability to Interpolate to a Subarea
Is there evidence of adverse effects under field conditions for water-dwelling organisms?	Various decision criteria related to zooplankton, fish, etc.	Moderate. Direct sampling of zooplankton and fish focused on achieving replication for representative areas, not spatial coverage of the entirety of the different lakes. Note that these organisms are mobile.	High for the lakes. There was adequate spatial coverage in the lakes and mobile organisms integrate exposure over time and space. Hazeltine Channel was not sampled for these endpoints.
Is there evidence of adverse effects under field conditions for sediment-dwelling organisms?	Quantitative – 20% reduction relative to reference	Moderate. Study designs focused on replication for representative areas.	Moderate to high. Multivariate statistical analysis looked for relationships between benthic endpoints and stressors. Recolonization experiment was particularly effective in highlighting that there is no obvious barriers to recolonization, but that re-establishment of the benthic community will be a relatively lengthy process.

5.0 CONCLUSIONS

5.1 Summary of Risk Conclusions

An ecological risk assessment was conducted to evaluate the ecological significance of the altered environmental conditions that are currently present in the study area as a result of the TSF embankment breach. Risk management and rehabilitation activities have been underway since the TSF embankment breach occurred and the physical impacts associated with the initial release are being addressed through a broad and consultative habitat rehabilitation program. This has been an adaptive program in which data collection and evaluation have been conducted over a period of three years to address a wide variety of investigation needs and regulatory requirements. The risk assessment provides a framework to evaluate the available data in light of the long-term management goal of successfully reintroducing a biologically diverse, functional, self-sustaining, and inter-dependent ecosystem.

The risk assessment is intentionally conservative in its approaches, assumptions, and decision criteria, consistent with relevant guidance and common practice in British Columbia. A problem formulation was completed to identify the extent of the problem and establish the objectives of the risk assessment. The problem formulation was summarized in a conceptual site model that highlighted stressor sources, receptors of interest, and potentially significant exposure pathways; the conceptual site model considers the site investigation data and our understanding of the physicochemical properties and fate of the COPCs. An exposure and effects assessment was conducted using a weight of evidence approach that evaluated a wealth of information to reach a conclusion about the magnitude of risks for each of the major ecosystem components. These conclusions for individual ecosystem components are provided in Sections 3.5 and 4.9 for the terrestrial and aquatic risk assessments, respectively. There are also several broad risk conclusions that apply to the study area as a whole, based on the totality of information considered:

- Copper is the primary COPC associated with tailings, irrespective of whether tailings were deposited to land or water. There were no other chemical constituents that were found to be a significant ecological risk that was attributable to the TSF embankment breach. The tailings also have structural and nutrient deficiencies that are important non-contaminant stressors.
- The magnitude of hazard associated with copper exposure was evaluated through comparison of the data to numerical standards and guidelines, but also included evaluation of specialized tests to evaluate leachability and bioavailability. There were numerous lines of evidence that showed that the copper associated with tailings has lower bioavailability than in the experiments used to derive those protective numerical guidelines and standards. This finding was consistent in sediment, water, and soil, and highlights that the magnitude of ecological risk is not equivalent to the magnitude of a screening level hazard quotient. A relatively low bioavailability does not mean that bioaccumulation is zero. Copper has bioaccumulated in different parts of the ecosystem (e.g., benthic invertebrates, fish, plants and soil invertebrates) at concentrations that are greater than reference areas.

- Several ecosystem components had the benefit of laboratory-based toxicity testing or other tools that allowed for experimental verification that the structural and nutrient deficiencies in the tailings are a significant risk. The specific attributes contributing to the deficiency varied somewhat by media (e.g., total organic carbon in sediment; plant available nutrients and bulk density in soil) but were broadly related to the fact that the tailings lack the heterogeneity and nutrient processing that is fundamental to a healthy soil or sediment ecosystem. This deficiency is expected to decrease over time as organic carbon and biological function returns to the system. Specific examples of the experimental verification include:
  - The side-by-side soil toxicity testing, which included a range of spiked and non-spiked copper concentrations in site soils and artificial controls
  - The manipulation of organic carbon concentrations in sediment toxicity testing
  - The benthic community recolonization experiment that involved a pair-wise transplantation of sediment between impacted and non-impacted profundal areas in Quensel Lake.
- A common approach in weight of evidence is to balance chemistry, toxicity, and bioaccumulation lines of evidence with field measurements. The chemistry, toxicity and bioaccumulation lines of evidence are well represented in the current risk assessment. Field-based lines of evidence are available, but constrained by study design, which can limit the ability to distinguish whether a response has occurred, the cause of any observed response, and the spatial extent of any such alteration. This is not surprising, given that the physical impacts associated with the TSF embankment breach were substantial and widespread.
- The balance of evidence from the experimental manipulations, field observations, and inferences based on case studies or relevant literature is that the adverse effects associated with the structural and nutrient deficiencies will be a larger influence on the long-term management goal of successfully reintroducing a biologically diverse, functional, self-sustaining and inter-dependent ecosystem than the influence of copper. The risks associated with copper are considered to be low; where risk levels were moderate, these were associated primarily with disruption to the physical substrate.

5.2 Application of Risk Conclusions to Remediation Areas

5.2.1 Summary Table

The following Table 52 provides an overview of the major risk conclusion for each of the remediation areas described in Figure 2. The risk conclusion provides an overall conclusion based on consideration of the various ecosystem components that were evaluated in the weight of evidence assessment. The balance of evidence with respect to identifying metals (e.g., copper) or physical factors (e.g. nutrient deficiencies) as a dominant causal factor for the observed impacts is also summarized. The overall trajectory in the field-based biological lines of evidence is also discussed. The causal factor and direction of change summary is intended to assist in linking this risk assessment to risk management planning.

Table 52: Summary of Risk Conclusions by Remediation Area

Area	Overall Risk under Current Conditions	Balance of Evidence for Causal Factor		Biological Direction of Change
		Contaminant	Physical	
Terrestrial				
Floodplain (Areas 2, 4, 5, 6, 7, 8)	Low to Moderate	Unlikely	Highly Likely	Slowly improving
Halo (Areas 2, 4, 5, 6, 7, 8)	Low to Moderate	Unlikely	Highly Likely	Improving
Aquatic				
Polley Lake (Area 3)	Low to Moderate	Unlikely	Likely	Improving
Hazeltine Channel (Areas 4, 5, 6)	Indeterminate	In progress	In progress	Improving
Edney Creek	Low to Moderate	Unlikely	Likely	Improving
Quesnel Lake Littoral (Area 8)	Low to Moderate	Possible	Likely	Improving
Quesnel Lake Profundal (Area 8)	Moderate	Possible	Highly Likely	Slowly improving
Quesnel River	Negligible	Not applicable	Not applicable	Not applicable

Overall risk ratings are presented as a range to reflect differences among ecosystem components or important spatial variation. This spatial variation is particularly relevant with respect to Area 8. See text for details.

5.2.2 Terrestrial

Tailings were deposited along the Hazeltine Channel in a pattern that was more amenable to separating risks based on lateral distance (i.e., floodplain versus halo) rather than by the risk management areas in Figure 2 (e.g., Polley Flats versus Lower Hazeltine Channel). Those areas were developed for remediation plan development and remain relevant for that purpose. The findings of the risk assessment are nevertheless relevant to the remediation areas because the risk assessment findings are logically linked to actions that will differ by area based on topography, ecosystem types and remedial habitat objectives. The detailed site investigation also concluded that soil contamination in tailings throughout the impacted area was part of the same population, which suggests that the appropriate approach is to determine risks associated with representative worst-case exposure concentrations rather than for specific worst-case exposure areas. Representative worst-case exposure concentrations were captured by multiple lines of evidence (e.g., invertebrate and plant tissue sampling; leachate testing, soil toxicity testing, PBET analysis) and interpreted using conservative approaches. The risk conclusions described in this report are equally applicable to all terrestrial remediation areas.

Overall, risks to the terrestrial environment are considered to be moderate. The physical nature of the tailings is expected to be an issue because tailings lack the heterogeneity and nutrient processing that is fundamental to a healthy soil ecosystem. This deficiency would be expected to decrease over time, particularly with physical intervention as has been taking place, as organic carbon and biological function returns to the system, but currently, the magnitude of the effect is such that a moderate conclusion is considered appropriate. There was no evidence that copper would act as an additional stressor over and above the influence of the physical and nutrient deficiency. Accumulation of copper in the food chain is considered to be low risk, based on the findings of the food chain model. The food chain model considered a wide variety of wildlife receptors with varying feeding strategies and dietary items and the general conclusion regarding low risks associated with copper bioaccumulation is therefore considered to be applicable to all wildlife species occupying the study area.

5.2.3 Aquatic

The tabular and narrative summaries provided in Section 4.9 are organized on the basis of broad receptor groups. Because the exposure to stressors following the TSF embankment breach varies as a function of distance and direction from the source, it is useful to summarize findings by spatial unit. In the case of aquatic risks, the major water bodies affected (Polley Lake, Hazeltine Channel, Quesnel Lake, and Quesnel River) provide the most logical basis for summarizing risks.

5.2.3.1 Polley Lake

The risks for most types of aquatic life are low in relation to other portions of the receiving environment. First, Polley Lake exhibits elevated concentrations of copper in reference conditions, which exceed the CSR Schedule 9 Standards ($SedQC_{TS}$) such that the concentrations of copper measured in 2014–2016 do not represent large increases relative to reference. In Polley Lake, no parameters were identified as stressors or COPCs in surface water based on chemistry data collected between March and August 2015. In terms of effects-based measures, there are multiple indications of recovery:

- Plankton—no adverse effects were observed in toxicity tests (acute or chronic tests with cladocerans) conducted in representative water samples, and no apparent change in biomass relative to background areas. The median concentration of copper in zooplankton collected from Polley Lake is lower than the upper limit of the normal range of samples collected from reference areas.

- Benthic Invertebrates—Polley Lake locations, in both deep and mid-depth locations, exhibited clear signs of biological recovery following the first year of sampling. Although there are indications of reduced density and diversity of biota, the magnitudes of these differences are small relative to other aquatic habitats. In 2014 and 2016, Polley Lake sediments indicated relatively few instances of significantly reduced survival or growth of either *C. dilutus* or *H. azteca*. Copper concentrations in benthic invertebrate tissue samples from Polley Lake, were significantly elevated relative to reference, but had lower concentrations than profundal Quesnel Lake.

- Fish—Polley Lake water quality did not exhibit acute or chronic toxicity to fish, with the exception of anomalous results for fathead minnows tested in the late fall of 2014 using pumped water samples. Median concentrations of copper in benthic invertebrate tissue are less than dietary threshold for adverse effects to fish. The age frequency information indicates a reduction in recruitment for rainbow trout following the TSF embankment breach, as well as a potential loss of the majority of the young-of-year for both rainbow trout and longnose sucker in 2014 and 2015.

- Piscivorous wildlife—Adverse effects are not expected.

5.2.3.2 Hazeltine Channel and Lower Edney Creek

The evaluation of risks for the Hazeltine Channel was particularly influenced by the timing of data collection and rehabilitation. Following the TSF embankment breach, Hazeltine Creek changes included erosion and translocation of the channel bed, erosion and downstream transport of portions of the riparian zone, removal of aquatic communities, and evidence of biological responses in sediment testing in early rounds of laboratory testing. However, given that construction and rehabilitation efforts (see for example Bronsro et al., 2016) have modified the site conditions, it is more informative to evaluate risks based on the current, post-rehabilitation conditions.

- Plankton—no data have been collected for this receptor group, as primary productivity has been assessed in Quesnel Lake at the mouth of the creek rather than within the creek itself.
- Benthic Invertebrates—There were statistically significant reductions (greater than 20% reduction) in diversity at upstream and downstream locations relative to reference for samples collected with Surber sampler or a kicknet. Data were limited to 2015, and creek rehabilitation was subsequently initiated. Edney Creek already shows clear evidence of recolonization, and similar recovery for the Hazeltine Channel is anticipated in response to the rehabilitation based on proximity to upstream organisms that provide a continued source of colonization. Results of previous toxicity testing indicated potential risk to macroinvertebrates (partly due to physical factors); however, these conditions are no longer present. Creek rehabilitation has reduced exposure and provided new habitat.
- Fish—Direct assessment of effects to fish in the Hazeltine Channel have not been evaluated, as monitoring has emphasized conditions in adjacent water bodies, and because fish have been excluded over the course of the creek rehabilitation. However, water quality monitoring indicates that conditions in the Hazeltine Channel are unlikely to be harmful to fish, as both total and dissolved copper concentrations have showed a progressive decrease from levels observed in the first few months of 2015, corresponding with changes in turbidity and TSS within the creek. Future monitoring of water quality in Hazeltine Creek is anticipated. Fish have been observed in the rehabilitated portion of Edney Creek and it is similarly expected that fish will re-occupy Hazeltine Creek once the fish barriers have been removed.
- Piscivorous wildlife—Adverse effects are not expected.

Overall, risks to aquatic receptors in Hazeltine Channel are indeterminate, but there is no reason to conclude that the observed copper concentrations would ultimately result in unacceptable risks to aquatic receptors based on the information obtained for other waterbodies during the course of the investigation. Future monitoring activities will include site-specific, effects-based measurements to confirm this hypothesis.

5.2.3.3 Quesnel Lake

The Quesnel Lake aquatic environment contains multiple habitats that reflect differences in both the limnology and in the degree of disturbance from the TSF embankment breach. As such, the distribution of effects and risks is highly variable. Risks are relatively low for organisms associated with the water column (plankton, pelagic fish). Water chemistry shows that copper and turbidity are unlikely to present hazards to aquatic life now that the initial water quality issues have subsided. However, impairment to the benthic community remains in large areas of the profundal environment where tailings accumulated, and also in shallow portions of the lake close to the Hazeltine Channel. These signs of impairment are associated with the physical burial of benthic communities and associated changes to substrates, which will require time to recover. The profundal areas are recovering, but the rate of recovery appears to be slower than in Polley Lake and the reconstructed Hazeltine Channel/Edney Creek channels.

- Plankton—There were no significant reductions in chronic water algal growth tests or vascular plant growth toxicity tests on field collected water samples, or apparent change in primary productivity. There were no chronic toxicity to cladocerans in representative non-turbid waters, and no discernable difference in copper concentrations of zooplankton tissue samples between the exposed area and the reference areas of Quesnel Lake. There was also no discernable response in total zooplankton biomass or abundance, or in the relative biomass or abundance of dominant taxa, relative to reference.
- Benthic Invertebrates— Mean copper concentrations of benthic invertebrates from the profundal near-field areas of Quesnel Lake were approximately 3-times higher than those from the profundal reference area. The results of toxicity tests with two species indicate potential risk, based on the observation of greater than 20% responses in several samples; however, results are likely influenced by physical factors, particularly TOC. Significant growth reductions have been observed for both test organisms in multiple sampling years. The responses are greatest in sediments for which there is a combination of high copper content (i.e., >600 mg/kg dw) and for which the TOC value is low (i.e., close to 0.1%). These factors are highly correlated and are diagnostic of tailings. Significant effects on invertebrate growth in the near-field sediment were eliminated with the addition of TOC or mixing the samples (50:50) with reference sediment. In terms of benthic community alterations, several Quesnel Lake assemblages remain impaired, as both near field and far-field deep water stations exhibit a statistically significant and greater than 20% reduction in mean invertebrate density in all three sampling years. Similarly, the near-field shallow Quesnel Lake station continues to exhibit a statistically significant reduction greater than 20% in mean density and diversity, and recovery is not yet evident. Overall, recovery at these stations is slower than what has been observed in Polley Lake.
- Fish—No meaningful impairment (i.e., performance is not inhibited by more than 25% relative to control samples) for fish toxicity tests including 7-d rainbow trout survival and growth and 7-d fathead minnow survival and growth toxicity tests on field collected water samples. Concentrations of copper in field-collected samples of diet (benthic invertebrates) are below dietary threshold for adverse effects. Information on fish histology is not indicative of an ongoing water quality-induced stress associated with the tailings. Condition factors of juvenile sockeye salmon collected from different sampling zones in Quesnel Lake do not indicate a tailings influence.
- Piscivorous wildlife—Adverse effects are not expected.

5.2.3.4 Quesnel River

There are fewer measurements of site-specific effects in Quesnel River relative to the other water bodies; however, the available evidence indicates a low magnitude of risk comparable to reference conditions:

- Plankton—No significant reductions in chronic water algal growth tests or vascular plant growth toxicity tests on field collected water samples. No chronic toxicity to cladocerans in representative non-turbid waters.
- Benthic Invertebrates—Benthic invertebrate samples collected from Quesnel River and the corresponding reference area in Cariboo River were more diverse than samples collected from the Hazeltine Channel, and exhibit concentrations of COPCs similar to reference conditions.

- Fish—No meaningful impairment observed (i.e., performance is not inhibited by more than 25% relative to control samples) for fish toxicity tests including 7-d rainbow trout survival and growth, 31-d rainbow trout survival and development, and 7-d fathead minnow survival and growth toxicity tests on field collected water samples.
- Piscivorous wildlife—Adverse effects are not expected.

5.3 Risk Monitoring Considerations

Risk assessment is inherently an iterative process. The risk conclusions in this report were based on a considerable amount of data collected over a three year span for a variety of project objectives. Notwithstanding the amount of available data, as with all risk assessments, there are areas of uncertainty identified during the course of the risk assessment. The following summary is intended to highlight topics that were identified during the course of the risk assessment for consideration in a future monitoring plan. These topics focus on instances where a data gap or uncertainty meant that a risk conclusion could not be reached, or when it was necessary to use an overly conservative assumption to supplement limitations in the available site-specific data. The points below are intended to resolve these areas of uncertainty. The main points described below have been further expanded on Table 53. Table 53 is intended to list specific items where further evaluation as part of a Comprehensive Environmental Monitoring Program should be considered.

5.3.1 Terrestrial

The main topics identified during the course of the terrestrial ecological risk assessment:

- Ongoing monitoring of the re-establishment of the soil community (microbial, plant and soil invertebrates) is appropriate. The early indicators with respect to root health and plant growth are positive, and the expectation is that the soil invertebrate community will re-establish itself as the soil structure and plant community improves over time.
- The wildlife food chain model was limited in terms of dietary concentrations for insectivores. There were no data available for flying insects and it was necessary to substitute a conservative assumption in lieu of site-specific data. The diversity and number of litter- and soil-dwelling invertebrate tissue samples was somewhat limited, reflecting that the soil invertebrate community is still in the process of being re-established. There is an opportunity for further refinement of the toxicity reference values for wildlife in conjunction with the refinement of the exposure concentrations.
- There were insufficient data to develop a quantitative risk estimate for amphibians and reptiles. There are a variety of lines of evidence which could be pursued to confirm that the residual concentrations of copper do not present an unacceptable risk. Further evaluation of monitoring data should be designed in light of the existing lines of evidence that show that copper bioavailability is low and that physical stressors such as habitat quality and physical constraints are more likely to be the primary driver of observed effects.

Table 53: Summary of Topics for Consideration in the Comprehensive Environmental Monitoring Program

Receiving Environment and Area	Receptor Group	Risk Conclusion	Key Uncertainty	Monitoring To Address Uncertainties (detailed in the CEMP)
Terrestrial – All areas	Plants	Low to moderate – see Table 22	Field observations of the plant community re-establishment is positive but in an early state of plant succession.	Field monitoring of plant performance will be incorporated as part of the remediation plan monitoring.
			Plant growth under-performed in soil tests, but the relative contribution of copper versus physical and structural limitations of the soil has not been confirmed.	Field monitoring of plant performance (see above). Focused investigations to determine relative contribution of copper versus physical limitations may be added, but only if needed based on findings from field monitoring.
	Soil invertebrates	Low to moderate – see Table 22	The early successional nature of soil conditions means limited soil invertebrates were available for sampling.	Field monitoring of soil invertebrate community will be incorporated as part of the remediation plan monitoring; Focused investigations to determine relative contribution of copper versus physical limitations may be added, but only if needed based on findings from field monitoring.
	Wildlife	Low – see Table 24	Existing mathematical relationships between soil and tissue concentrations are based on currently available data. This results in uncertainty in how risk estimates are being applied to smaller areas.	Field monitoring of plants and soil invertebrate (see above) will include sampling for tissue chemistry to supplement the currently available data.
			Understanding of copper bioavailability was informed by available PBET data and literature values, resulting in uncertainty in the HQ.	Field monitoring (see above) will include collection of soil samples for PBET analysis to increase spatial coverage and diversity of soil types covered by PBET. Targeted study to expand PBET analysis to determine relative bioavailability (e.g. include additional types of items in the PBET analysis).
			Toxicity Reference Values (denominator in the HQ calculation) were based on screening level benchmarks.	No specific field activities. Desktop refinement to develop a site-specific toxicity reference value to replace the default TRVs.

Receiving Environment and Area	Receptor Group	Risk Conclusion	Key Uncertainty	Monitoring To Address Uncertainties (detailed in the CEMP)
Terrestrial – Riparian Habitats	Amphibians	Indeterminate	Amphibians have not been formally assessed beyond the qualitative observations described in the risk assessment.	Existing data (potentially with a targeted supplemental program) to be used to determine a screening-level estimate of hazards to amphibians exposed to soil, sediment and water.
Aquatic – Hazeltine Channel	Benthic invertebrates and fish	Indeterminate – effects-based lines of evidence were deferred while restoration was in progress	Concentrations of copper exceed water quality guidelines (i.e., there is a potential hazard).	Monitoring (post-habitat rehabilitation) to evaluate benthic invertebrate re-colonization. Ongoing monitoring of Hazeltine Creek water chemistry to evaluate trends.
Aquatic – Quesnel Lake	Benthic invertebrates	Low to moderate – see Table 49	Toxicity test performance was reduced but relative contribution of copper versus physical and structural limitations of the sediment has not been confirmed. Diversity metrics show improvement, transplant studies show positive recolonization potential but abundance remains reduced.	Monitoring of benthic community recovery to confirm that recolonization is progressing and that copper concentrations are not a barrier.
Aquatic – Polley Lake	Fish	Low – see Table 50	Clarification of the cause of trout age-class structure alteration in Polley Lake.	Evaluate age class frequency of fish species in Polley Lake. Monitor copper concentrations in water samples from Polley Lake.

5.3.2 Aquatic

Topics identified during the course of the aquatic ecological risk assessment:

- Activities for Hazeltine Creek over the last three years have focused on rehabilitation and restoration. Water quality subsequent to the rehabilitation has improved and stabilized, albeit with occasional exceedances of ambient water quality guidelines. Post-rehabilitation monitoring to confirm that the ecological function of Hazeltine Creek has been re-established is appropriate, in conjunction with water quality monitoring.
- Progress in recolonization of lake sediment by benthic organisms should be monitored. This could include direct measurement of diversity and abundance in benthic organisms for those waterbodies where recolonization appears to be well underway (e.g., Polley Lake), or an extension of the existing transplant study for areas in Quesnel Lake where progress is expected to be slower. A key point with respect to Quesnel Lake is that recolonization is also a function of distance from a source population—direct monitoring without considering the influence of distances is not appropriate.

6.0 CLOSURE

The design and execution of investigations completed by Golder to support this ecological risk assessment was directed by Ms. Trish Miller, M.Sc., R.P.Bio., and Dr. Reidar Zapf-Gilje, P.Eng. A draft problem formulation and preliminary risk assessment was prepared in January 2016 under the supervision of Ms. Miller, Dr. Zapf-Gilje and Mr. Gary Lawrence, MRM, R.P.Bio. This current problem formulation and risk assessment was prepared by Mr. Blair McDonald, MET, R.P.Bio working with Ms. Miller, Mr. Lawrence, Dr. Jordana van Geest, Ph.D., R.P.Bio., Ms. Arainn Atkinson, B.A., E.P.t., and Ms. Barbara Wernick, M.Sc., R.P.Bio. to incorporate additional information gained from supplemental investigations conducted throughout 2016, as well as feedback provided by the Technical Working Group and its contracted reviewers on the draft problem formulation and preliminary risk assessment document. This current submission (with its partner documents for the human health risk assessment and the combined technical appendices) supersede the January 2016 draft submission.

Mr. McDonald, Ms. Miller, and Dr. Zapf-Gilje have demonstrable experience in conducting ecological risk assessments pertinent to the purposes of the Contaminated Sites Regulation. The risk assessment was conducted in general accordance with Ministry-approved protocols, guidance, procedures, policies, methods and standards of professional practice, and the information used in the performance of the risk assessment and the conclusions of the risk assessment reported herein are true and accurate based on the current knowledge as of the date completed. All three senior risk assessment practitioners are Contaminated Sites Approved Professionals (Risk) who have been qualified to make recommendations to a Director with respect to issuance of risk-based Certificates of Compliance under the Contaminated Sites Regulation. Mr. Lawrence and Ms. Wernick are also senior practitioners with demonstrable experience in implementing weight-of-evidence approaches for ecological risk assessment and other applications.

A summary of the qualifications for Golder scientists who have had substantive involvement in this project is provided in Appendix Q. Golder acknowledges the contribution of numerous third-party subject matter experts and their project teams who have contributed to the technical deliverables that form the basis of this weight of evidence assessment, including Dr. Suzanne Simard, R.P.F. (UBC Department of Forestry), Mr. Pierre Stecko, M.Sc., R.P.Bio. (Minnow), Mr. Steve Day, P.Geo. (SRK Consulting), and Dr. Chris Kennedy, P.Geo. (formerly SRK Consulting, now AgnicoEagle Mines). Much of the sampling, data verification and management was carried out by staff from MPMC's Environmental Department.

Readers are directed to the Statement of Limitations which follows the text and forms an integral part of this report.

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STUDY LIMITATIONS

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