2021 Cigar Lake Operation Environmental Risk Assessment Summary

Executive Summary

Cameco Corporation's 2021 Environmental Risk Assessment (ERA) for the Cigar Lake Operation was completed according to the CSA N288.6-12 standard for Environmental Risk Assessments at Class I Nuclear Facilities and Uranium Mines and was consistent with previous assessments conducted. This document summarizes the ERA and demonstrates the compliance with CSA N288.6-12.

The Human Health Risk Assessment concluded that *Constituents of Potential Concern (COPCs) related to the Operation are not expected to present a risk. Although COPC concentrations are predicted to increase in Seru Bay as a result of treated effluent release, only slight changes in water quality are expected within Seru Bay and minimal to no changes to water quality are expected outside Seru Bay. Human health and environmental receptors remain protected in the vicinity of the Operation. Further, the ERA and routine monitoring results continue to demonstrate that the site remains within the objective of the licensing basis and previous environmental assessment predictions.*

Introduction

Cameco Corporation (Cameco) owns (54.5%) and operates the Cigar Lake Operation (the Operation) (Figure 1). It is located in northern Saskatchewan, at the eastern edge of the Athabasca basin at the south end of Waterbury Lake, approximately 660 km north of Saskatoon. Development of the Cigar Lake mine began in 1981, and commercial operation began in May 2015. Milling of the Cigar Lake ore is being conducted at the McClean Lake site operated by Orano Canada Inc.

Objectives

An Environmental Risk Assessment (ERA) is a systematic process used to identify and assess the potential risk posed by releases from the Operation to people and the environment. ERAs follow general guidance provided by CSA and various agencies, such as Health Canada (HC), Environment and Climate Change Canada (ECCC), Canadian Council of Ministers of the Environment (CCME) and the Canadian Nuclear Safety Commission (CNSC).

Figure 1. Cigar Lake Operation

There are two parts to an ERA – 1) an assessment of the exposure and potential risk to people who use the area through a human health risk assessment (HHRA) and 2) an assessment of living things in the environment (such as plants, insects, animals) through an ecological risk assessment (EcoRA).

The Cigar Lake ERA was used to *"...characterize the [potential] risk posed by contaminants and physical stressors in the environment on biological receptors, including the magnitude and extent of the potential effects associated with [the] facility* (CSA 2022, p.12)*."* In addition, the conclusions of the current ERA were compared to those provided in the most recent environmental assessment completed for the Operation, the 2011 Environmental Impact Statement (EIS).

Overview of ERA Approach

The first step in conducting an ERA (Figure 2) is to detail the releases from the Operation and to understand how they move in the natural environment. Environmental monitoring is conducted at the Cigar Lake Operation in accordance with the CSA N288.4-19 (Environmental Monitoring Programs at Nuclear Facilities and Uranium Mines and Mills) and CSA N288.5-22 (Effluent and Emissions Monitoring Programs at Nuclear Facilities) and N288.7-15 (Groundwater Protection Programs at Class 1 Nuclear Facilities and Uranium Mines and Mills) standards, along with requirements established in the provincial Approval to Operate, CNSC Environmental Management Program, and ECCC Environmental Effects Monitoring programs. Data collected through these monitoring programs at the Operation help to inform this step.

Once the releases are understood, the relevant COPCs need to be identified. This is a list of the key radiological and non-radiological constituents released to air and water from site operations. It is developed from knowledge of the facility, environmental monitoring data, and feedback from regulators, community members and other stakeholders. In developing the list of COPCs, some constituents are removed from further consideration if they are released in very small quantities, if they are present at or below natural background levels, or if they are determined not to be a concern from a human or ecological health perspective. The concentration(s) of COPCs in the environment (e.g., soil, surface water, air) are determined in the natural areas near the Operation using monitoring data, modelling, or a combination of both.

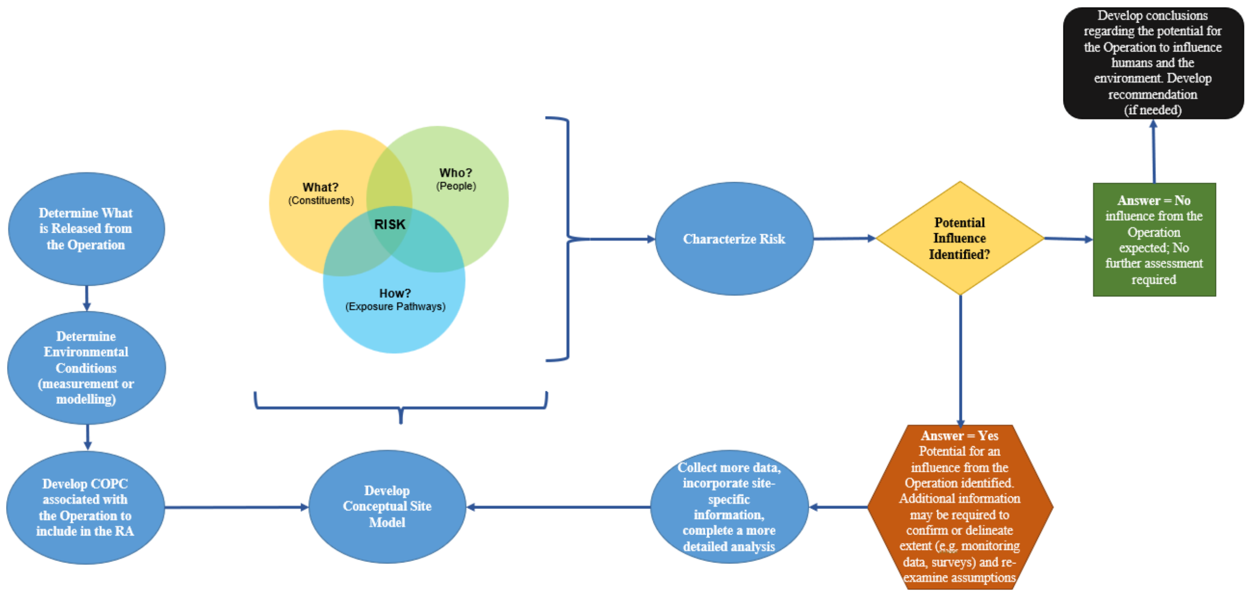


Figure 2. Environmental Risk Assessment Overview

The foundation of the risk assessment is the Conceptual Site Model (CSM). The CSM summarizes how the COPCs are released and are expected to move in the environment, as well as identifies who uses the land, including both people and biota (wildlife, plants). This information, together with information on the potential influence of COPCs, are used in the risk assessment. The pathways assessment (also called risk characterization or risk assessment) uses information on What (selected COPCs), Who (identify receptors) and How (exposure pathways) to assess the risk.

The CSA standard N288.6-12 provides a systematic approach and calculations that are used to estimate the exposure of the human or ecological receptor to each of the COPCs. The calculations estimate the uptake of COPCs from the different environmental media and indicate how the COPCs are passed up the food chain. A cautious approach is taken in the assessment using conservative assumptions that are likely to overestimate the exposure. An example of a conservative assumption can be seen regarding the home ranges of the evaluated species. Those species with larger home ranges, such as wolf, moose and woodland caribou, are conservatively assumed to spend a significant amount of time in the exposure area; however, it is expected that they would range over a larger area.

Potential changes in the environment are determined using a weight-of-evidence approach. One part of this is to calculate a screening index (SI). In simple terms, an SI is the concentration or exposure level divided by published scientific benchmarks that have been deemed unlikely to adversely influence the receptor (Figure 3). These benchmarks can come from research or field studies, regulatory standards and objectives, scientific literature or other credible sources. If no potential changes are identified (i.e., if SI is less than 1), then influences on the environment are not expected. Due to the cautious nature of the calculations, an SI greater than 1 indicates that further assessment may be required. This can include more detailed analysis, additional field data and site-specific information.

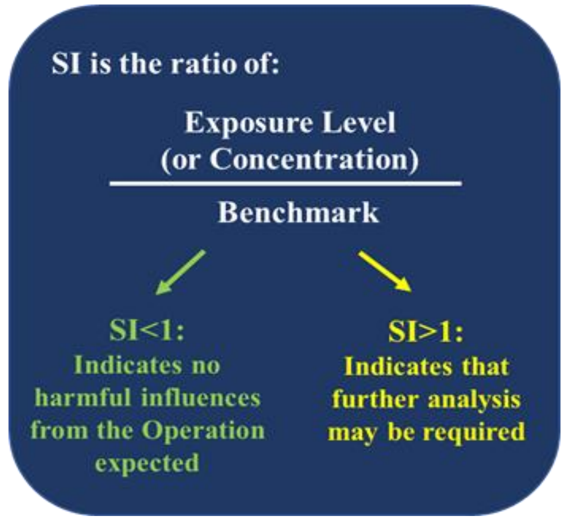


Figure 3. Screening Index (SI)

In a weight-of-evidence approach, all information is considered to reach an overall conclusion on the potential for a response. For example, for the assessment of aquatic insects that are in sediment, the SI will be considered along with information on the type of insects and how many are present. Once the assessment is complete, a conclusion regarding the potential harm to people or the environment is developed.

The following sections provide more information specifically about the Cigar Lake Operation, the releases into the environment from the Operation, selection of COPCs and receptors, pathway characterization, and results and conclusions of the ERA. The input from the local communities is also highlighted. For example, ecological receptors were selected based on surveys completed in the Operation area as well as other considerations including local resource user interviews and input from local communities.

Site Description

The Cigar Lake Operation is situated near the southern shore of Waterbury Lake between the Aline Creek and Cigar Creek drainages (Figure 4). The Aline Creek system flows into Seru Bay of Waterbury Lake, while the Cigar Creek system flows into Longyear Bay of Waterbury Lake. The aquatic environment study areas considered in the ERA incorporate all of Waterbury Lake, including Seru Bay and Longyear Bay. The terrestrial environment study area includes a 100 km^2 area centered on the Operation.

The nearest permanent community is Wollaston Lake, which is located approximately 80 km east of the site.

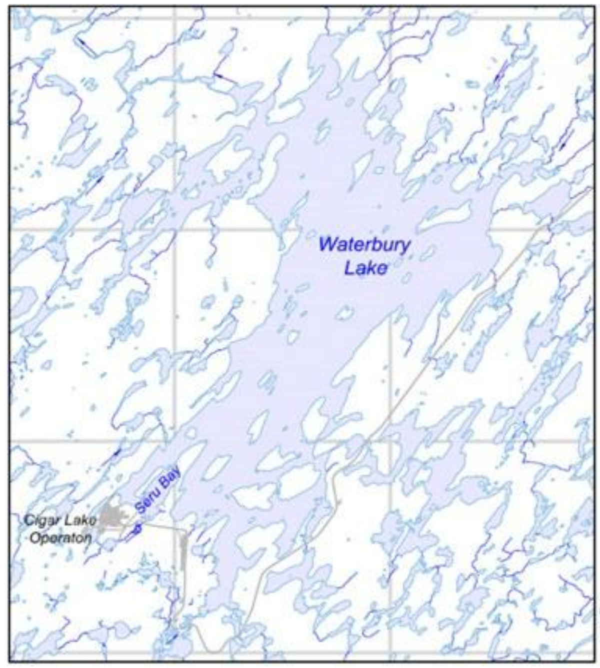


Figure 4. Cigar Lake

Facility

The Cigar Lake Operation itself includes underground mining, waste rock stockpiles, settling ponds, Mine Water Treatment Plant (MWTP) effluent treatment facility, camp infrastructure, garbage disposal landfill, and sewage lagoon. There is no tailings management facility at the Cigar Lake Operation, since all mined ore is processed at the McClean Lake Operation mill, located approximately 80 km away. The majority of ore processing equipment is located underground at the Cigar Lake Operation, which gives the site a small surface footprint.

Releases into the Aquatic Environment

Prior to July 16, 2013, treated water from the MWTP was released to a muskeg area southwest of the site and from there into Aline Lake. Beginning August 7, 2013, water from the Operation is treated and released through a multi-port diffuser into Seru Bay of Waterbury Lake. Detailed modelling, completed as part of the 2011 EIS, was conducted to evaluate water and sediment quality in the environment downstream from the Operation.

The amount and quality of treated water released for use in the ERA were based on the measured data from the MWTP at the site and on an understanding of potential future volumes. Two scenarios were considered for the treated effluent release: an Expected Loading scenario, which represents the current estimate of future effluent flows and concentrations; and a more conservative Upper-bound Loading scenario. To investigate the potential influence of non-routine discharge from inflow events at the Operation, sensitivity case scenarios were also run. These sensitivity cases included a non-routine discharge to each of the Expected and Upper-bound Loading scenarios.

The assessment also considered the potential long-term groundwater loads from various surface sources at the Operation. Groundwater is expected to reach Cigar Lake, Longyear Bay of Waterbury Lake, Aline Lake, and Aline Creek, which flows to Seru Bay of Waterbury Lake.

The movement of COPCs in the environment was modelled using a computer program called ADEPT (Assessment of the Dispersion and Effects of Parameter Transport), which is a contaminant dispersion and transport model for waterbodies that includes pathways and risk assessment calculations. The model can assess a variety of COPCs and considers numerous lakes/rivers/wetlands/bays and multiple branches of a watershed. As expected, as the Operation proceeds, water and sediment concentrations are predicted to increase. Once the site is decommissioned and treated effluent is no longer released, the concentrations are expected to decline and return to pre-operational conditions. The long-term groundwater loads from surface sources are expected to have minimal influence on the receiving environment.

Releases to the Atmospheric Environment

In accordance with the Saskatchewan Air Quality Modelling Guideline, air dispersion modelling was used to evaluate the influences of the Operation on ambient air quality over the life of the mine. The releases from the facility include mine ventilation, waste rock storage, and road dust. The CALMET/CALPUFF modelling package was used to predict concentrations of various COPCs. Overall, it was predicted that the Operation would have only a limited influence on air quality. Exceedances of Saskatchewan air quality guidelines are conservatively estimated to only extend to within 500 m from the project footprint or mine access road for particulate matter and NO_2. Within five kilometres of the Operation, all COPCs concentrations are predicted to return to near background levels.

Selection of COPCs

The selection of COPCs compared the effluent quality data from the final point of control before discharge to the Saskatchewan Environmental Quality Guidelines for protection of aquatic life. Other considerations in the selection of COPCs included activities at the site (identifying air quality parameters), requirements for site monitoring, as well as alignment with the COPCs identified in the 2011 EIS. The list of COPCs selected for the assessment include:

- Metals (and metalloids): arsenic, cobalt, copper, lead, molybdenum, nickel, selenium, uranium, and zinc.
- Radionuclides: uranium-238, lead-210, polonium-210, radium-226, and thorium-230
- Total Dissolved Solids (TDS) was included as it represents inorganic salts present in solution in water including calcium, magnesium, sodium, and potassium cations and carbonate, bicarbonate, chloride, sulphate, and nitrate anions.
- Other general chemistry constituents selected for inclusion in the COPCs list are ammonia, chloride, nitrate, and sulphate.
- Additional COPCs selected for inclusion for air quality are total suspended particulates (TSP) and constituents, particulate matter (PM_{10}, $PM_{2.5}$), gaseous COPCs (nitrogen oxides (NO_x), sulphur dioxide (SO_2), and radon (Rn-222).

Local Environment

The Cigar Lake Operation is located in the Athabasca Plain ecoregion of the Boreal Shield ecozone. The climate in this region is typical of the continental sub-arctic region and is characterized by short, cool and moist summers, and very cold, dry winters. This ecozone is classified as having a subhumid high boreal climate. The average frost-free period is approximately 90 days (Minister of Supply and Services Canada 1996).

Glacial till is the dominant surficial deposit at the Cigar Lake Operation area and is the primary constituent of the ground moraine and drumlin landforms that characterize the region. Soils in the area are dominantly brunisolic in well-drained sites. Brunisolic soils are boreal forest soils that develop in sandy glacial parent material under jack pine forests (University of Saskatchewan 2009).

The Cigar Lake Operation lies southwest of Waterbury Lake within the Waterfound River basin, part of the larger Mackenzie River basin. Waterbury Lake is the headwater for the Waterfound River, which is a tributary of the Fond du Lac River.

Human Health Risk Assessment

HHRA Problem Formulation

Human receptors, representing a range of people who may live and work in the study area were included in this assessment. The selected human receptors are consistent with those from the 2011 EIS and include an adult working at the Operation's camp (e.g., cook, security) and hypothetical receptors including a family (adult, child, toddler) working four months a year at the Waterbury Lodge. Input from stakeholder interviews and the local users was important for defining the appropriate scenarios. Local trappers have reported fishing in the Cigar Lake area and using the local environment. The hypothetical Waterbury Lodge Worker receptor was defined to be more connected to the environment surrounding the Cigar Lake Operation than the local trapper would be and thus any potential risks to the trapper are included in assessment of the lodge worker.

In determining characteristics for the human receptors, the objective was to make conservative but realistic assumptions drawing from knowledge of local land use (e.g., interviews with local trappers) as well as regional information, such as the dietary survey of the Hatchet Lake Denesuline First Nation.

The consumption of traditional food was based on a dietary survey of the Hatchet Lake Band done in 2000. This survey provides information on traditional food intakes for the Wollaston Lake community. More recent dietary surveys completed for the communities of Black Lake and Stony Rapids and Fond du Lac indicate overall lower rates of meat consumption, particularly for caribou compared with the 2000 Hatchet Lake Band study. The rates are also higher than the Boreal Shield diets provided in the First Nation Food, Nutrition and Environment (FNFNES) survey for Saskatchewan (Chan et al. 2018). It is speculated that the overall higher rates of traditional foods consumed in the 2000 Hatchet Lake Band study compared with more recent studies is in part due to store bought foods being far less accessible to the region 20 years ago.

While the survey found that barren-ground caribou are hunted and form a part of their diet, it was indicated that the Hatchet Lake community largely hunts caribou in the area to the north and northeast of Wollaston Lake, an area not within the potential influence of the Cigar Lake Operation. The rest of the traditional food while in the area was assumed to be taken from local sources.

Other characteristics were taken from typical sources such as drinking water intakes and body weights were taken from Richardson (1997) and soil ingestion rates were from Wilson et al (2013).

Consistent with N288.6-12, the receptor exposure pathways for the human health assessment include drinking water, soil contact and incidental ingestion, inhalation, external radiation and food obtained from local sources (meat, fish, berries, and other traditional food consumption).

A CSM is a representation of the biological, physical and chemical processes that determine the ways that constituents move from sources through the environmental media to environmental receptors. Figure 5 presents the CSM for the Operation, including pathways considered in the EcoRA.

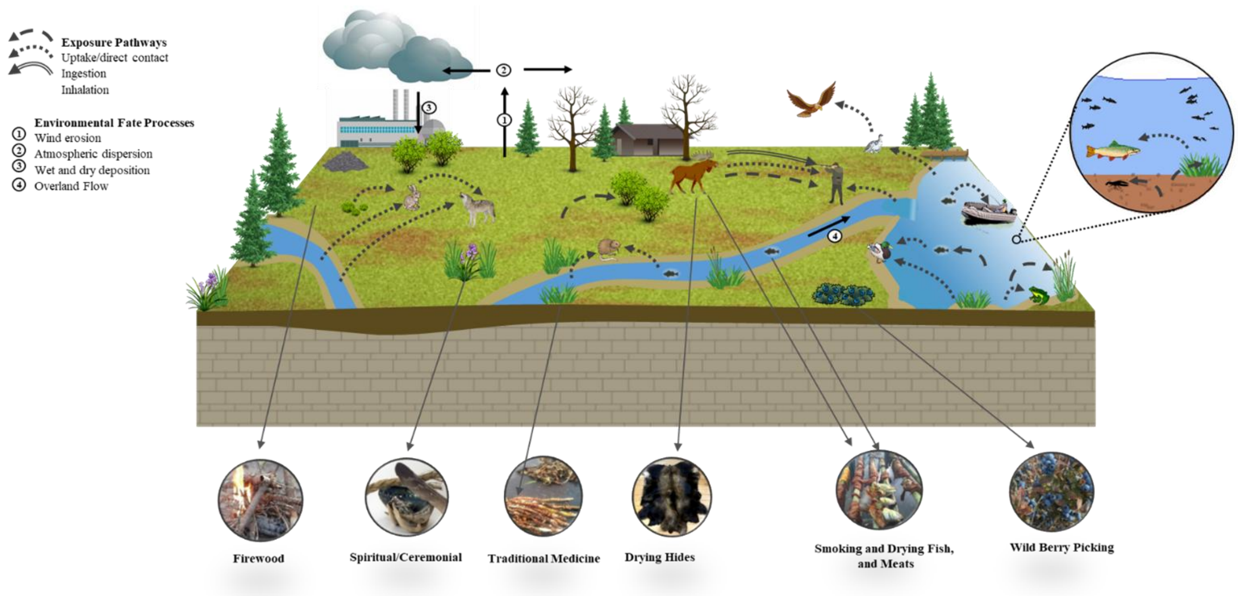


Figure 5. Cigar Lake Operation conceptual site model

There is uncertainty associated with the diet assumptions used in the assessment, since there is a wide range of consumption values reported in communities across Canada. Distributions were applied to human exposure assumptions, in order to capture variability between individuals and acknowledge the uncertainty in the assumptions. Minor exposure pathways, such as indirect exposure from firewood, was not included. However, this is not expected to affect the conclusions of the assessment as this is a minor exposure pathway compared to dietary intakes. One of the human receptor groups considered in the assessment (Waterbury Lake Lodge operator) was defined for a hypothetical future situation. Currently, no one is operating a lodge on Waterbury Lake, and so characteristics of this hypothetical human receptor group were made based on conservative land use assumptions, but are based on assumptions nonetheless.

HHRA Exposure Assessment

In this step of the HHRA, the potential exposure to COPCs is estimated. This needs to consider factors like concentrations, duration, and frequency of exposure. The concentrations in the environment are primarily estimated using modelling, which is described later in this section. The model includes comparison to available measured data to make sure that the concentrations are reasonable.

The exposure duration and frequency for the selected receptors:

- An adult working for six months a year (two weeks on / two weeks off rotation) at the Cigar Lake Operation camp (e.g., cook, security), obtaining water, air, berries, and fish from Longyear Bay during the operational and decommissioning period while at the camp.
- A family, including a range of receptor age groups (adult, child, and toddler), residing for four months a year while working at the Waterbury Lodge (hypothetical receptors) during the operational and decommissioning period and the post-decommissioning period. While in the area they obtain water, air, berries and medicinal plants locally and hunt and fish for food to consume throughout the year.

Dermal exposure to soil and water was also considered and followed the approach outlined by Health Canada (2010a) for soil and U.S. EPA (2004) for water, in accordance with N288.6-12 (CSA 2012). The typical values for exposed skin surface area (SA) and soil loading to skin (SL) by human receptor type were taken from Health Canada (2010b). Relative absorption factors for skin (RAFskin) for various COPCs in soil are provided in Health Canada (2021) and permeability coefficients (Kp) for inorganics in water are provided in U.S. EPA (2004). Exposure frequency (EF) is typically assumed to be 1 event per day for soil contact. Fraction of time spent swimming (OFw) and bathing (OFw') are based on site-specific assumptions. The default value for fraction of time swimming (OFw) is 0.014 for adults, teens, children, and toddlers, based on a one-hour swim per day during four summer months; infants are not assumed to swim (CSA 2020). The default value for fraction of time bathing (OFw') is 0.014 for all age groups, based on a 20-minute bath per day (CSA 2020). The ADEPT model assumes that swimming occurs in natural waterbodies and does not consider exposure to swimming pool water.

Receptor intakes through ingestion are estimated using the equation (1). This equation applied to all age groups.

$$I_f = C_w \times WIR + C_s \times SIR + C_{tea} \times TIR + \sum_v C_v \times VIR_v + \sum_k C_k \times FIR \times F_k \qquad (1)$$

Where:

I_f	=	COPC intake by human receptor (g/yr or Bq/yr)
C_w	=	COPC concentration in water (g/m^3 or Bq/m^3)
WIR	=	Water ingestion rate for human receptor (m^3/yr)
C_s	=	COPC concentration in soil (g/g (dw) or Bq/g (dw))
SIR	=	Soil ingestion rate for human receptor (g (dw)/yr)
C_{tea}	=	COPC concentration in medicinal tea (g/g (ww) or Bq/g (ww))
TIR	=	Tea ingestion rate for human receptor (m^3/yr)
C_v	=	COPC concentration in vegetation (g/g (ww) or Bq/g (ww), v=berries, medicinal plants
$VIR_{,v}$	=	Vegetation ingestion rate for human (g (ww)/yr)
C_k	=	COPC concentration country foods, k = such as fish, moose, etc. (g/g (ww) or Bq/g (ww))
FIR	=	Food ingestion rate for human receptor (g (ww)/d)
$F_{,k}$	=	Fraction of human receptor diet comprised of k, for k = country food (-)

The intake for human receptors from inhalation of air is calculated following equation (2).

$$I_{inh} = C_a \times I \qquad (2)$$

Where:

I_{inh}	=	COPC intake by inhalation (g/yr or Bq/yr)
I	=	Inhalation rate (m^3/yr)
C_a	=	COPC concentration in air (g/m^3 or Bq/m^3)

The calculation of exposures for specific human receptors considered in an assessment applies site-specific exposure assumptions to the calculations in equations (1) and (2), as illustrated in equation (3).

$$C = (C_{Loc-1} \times F_{Loc-1}) + (C_{Loc-2} \times F_{Loc-2}) + (C_{BG} \times (1 - F_{Loc-1} - F_{Loc-2})) \quad (3)$$

$$\begin{Bmatrix} Water \\ Soil \\ Air \\ Tea \\ Vegetables \\ Berry \\ Fish \\ Meat \end{Bmatrix} \qquad \begin{Bmatrix} Water \\ Soil \\ Air \\ Vegetables \\ Berry \\ Fish \\ Meat \end{Bmatrix} \qquad \begin{Bmatrix} Water \\ Soil \\ Air \\ Tea \\ Vegetables \\ Berry \\ Fish \\ Meat \end{Bmatrix}$$

A number of pathways are considered specifically for radiological dose; consistent with N288.6-12 (CSA 2012), these include exposure to radon, immersion in air, immersion in water, and groundshine. Dose from contaminated shoreline sediment (Clause 6.3.4.2.10, CSA 2012) is not specifically considered in the ADEPT model, as this is negligible.

Dose to human receptors from radon is calculated following equation (4). The value for F, degree of equilibrium between radon and decay products, is assumed to be the default value of 0.4 (Clause 6.3.4.2.11, CSA 2012).

$$Dose_{Rn} = \left[\frac{C_{Rn}}{3700}\right] \times F \times \left[\frac{ET}{170}\right] \times 4 \quad (4)$$

Where:

$Dose_{Rn}$	=	COPC dose from radon (mSv/yr))
C_{Rn}	=	Radon concentration in air (Bq/m^3)
F	=	Degree of equilibrium between radon and decay products (-)
ET	=	Exposure time (hrs/yr)
3700	=	Conversion factor for radon (Bq/m^3/WLM)
170	=	Conversion factor for time (hrs/WLM)
4	=	Conversion factor for WLM (mSv/WLM)

Dose to human receptors from immersion in air is calculated following equation (5).

$$Dose_{imm} = f_o \times [f_u + (1 - f_u)S_b] \times DC_a \times C_a \times 1000 \quad (5)$$

Where:

$Dose_{imm}$	=	Dose from immersion in contaminated air (mSv/yr)
f_o	=	Fraction of total time spent by an individual at a particular location (-)
f_u	=	Fraction of time spent outdoors at a particular location (-)
S_b	=	Building shielding factor (-)
DC_a	=	Effective dose coefficient for a semi-infinite cloud (Sv/yr per Bq/m^3)
C_a	=	COPC concentration in air (Bq/m^3)
1000	=	Conversion factor (mSv/Sv)

Equation (5) is based on Clause 6.3.4.2.4 of N288.6-12 (CSA 2012), which is based on guidance provided in N288.1-20 (CSA 2020). Fraction of time at the location (fo) is based on site-specific assumptions, while fraction of time outdoors (fu) is assumed to be 0.2, based on guidance provided in Clause 6.2.4 of N288.1-20 (CSA 2020). The recommended default value of 0.5 for building shielding (Sb) is used for the assessment, although it is acknowledged that this value depends on many things, such as type and size of the building, wall and roof thickness, and location of the receptor within the building. U.S. EPA (2019, Table 4-6) provides guidance for effective dose coefficients for a semi-infinite cloud (DCa).

Dose to human receptors from immersion in water is calculated following equation (6).

Equation (6) is based on Clause 6.3.4.2.6 of N288.6-12 (CSA 2012) and guidance for input values is provided in N288.1-20, Clause 6.16.1 (CSA 2020). The default value provided for the correction factor to account for the finite size of a bathtub (Dc) is 0.7 (CSA 2020). The removal factor (ρ) is conservatively assumed to be 1 (CSA 2020). U.S. EPA (2019, Table 4-7) provides guidance for effective dose coefficients for immersion in an infinite uniformly contaminated water medium (DCwi).

$$Dose_{wi} = (OF_w + D_c \times \rho \times OF_w' + \rho \times OF_w'') \times DC_{wi} \times C_{wi} \times 1000 \quad (6)$$

Where:

$Dose_{wi}$	=	Dose from immersion in contaminated water (mSv/yr)
OF_w	=	Fraction of year spent swimming in a surface waterbody (-)
D_c	=	Correction factor to account for the finite size of a bathtub (-)
ρ	=	removal factor to account for processes such as sedimentation and removal of radionuclides by water treatment plants (-)
OF_w'	=	Fraction of year spent bathing (-)
OF_w''	=	Fraction of year spent swimming in swimming pool (-)
DC_{wi}	=	Dose coefficient for immersion in an infinite uniformly contaminated water medium (Sv/yr per Bq/m^3)
C_{wi}	=	COPC concentration in water (Bq/m^3)
1000	=	Conversion factor (mSv/Sv)

Dose to human receptors from groundshine is calculated following equation (7).

$$Dose_g = f_o \times f_r \times [f_u + (1 - f_u)S_g] \times DC_g \times C_g \times \rho_b z_{soil} \times 1000 \quad (7)$$

Where:

$Dose_g$	=	Dose from contaminated ground deposits (groundshine) (mSv/yr)
f_o	=	Fraction of total time spent by an individual at a particular location (-)
f_r	=	Dose reduction fraction to account for non-uniformity of the ground surface (-)
f_u	=	Fraction of time spent outdoors at a particular location (-)
S_g	=	Groundshine shielding factor by buildings (-)
DC_g	=	Effective dose coefficient for an infinite plane ground deposit (Sv/yr per Bq/m^2)
C_g	=	COPC concentration in ground surface (assumed to be C_s) (Bq/g)
P_b	=	Soil dry bulk density, depends on soil type (g (dw)/m^3). See soil model.
Z_{soil}	=	Depth of top mixed soil layer (m) (0.20 m recommended, CSA 2020)
1000	=	Conversion factor (mSv/Sv)

Equation (7) is based on Clause 6.3.4.2.9 of N288.6-12 (CSA 2012) and Clause 6.14.1 of N288.1-20 (CSA 2020), with soil dry bulk density and soil layer depth applied for units conversion (Bq/m^2 to Bq/g). Guidance for input values is provided in N288.1-20, Clause 6.14.1 (CSA 2020). The default value provided for the dose reduction fraction to account for non-uniformity of the ground surface (fr) is 0.7 (CSA 2020). Fraction of time at the location (fo) is based on site-specific assumptions, while fraction of time outdoors (fu) is assumed to be 0.2, based on guidance provided in Clause 6.2.4 of N288.1-20 (CSA 2020). The default value for building shielding factor for groundshine (Sg) is 0.7, based on Clause 6.14.3 of N288.1-20 (CSA 2020). U.S. EPA (2019, Table 4-1) provides guidance for effective dose coefficients for an infinite plane ground deposit (DCg).

Dose coefficients are used to estimate radiological dose, these were taken from ICRP (1996).

The ADEPT model is used to help understand how releases from the Operation may influence water and sediment quality. The environment is broken down to different compartments (e.g., Seru Bay is divided into two sections), and the water and sediment quality is tracked in each compartment within ADEPT. Each compartment must be described, for example, the water depth, area and active sediment layer. The model considers how COPCs can move between compartments (river flow, lake currents), how it can settle from water to sediment, exchange with sediment, as well as other processes. The model can then predict how the water quality is expected to change over time. ADEPT is written in a programming language called MATLAB, which is commonly used for this kind of scientific work.

The model takes into account that certain input parameters, (e.g. precipitation) change from year-to-year and that precise estimates are not always known. Rather than using just one value, a range of possible values can be evaluated through the use of a probabilistic framework. This helps us understand how confident we can be in the model's predictions.

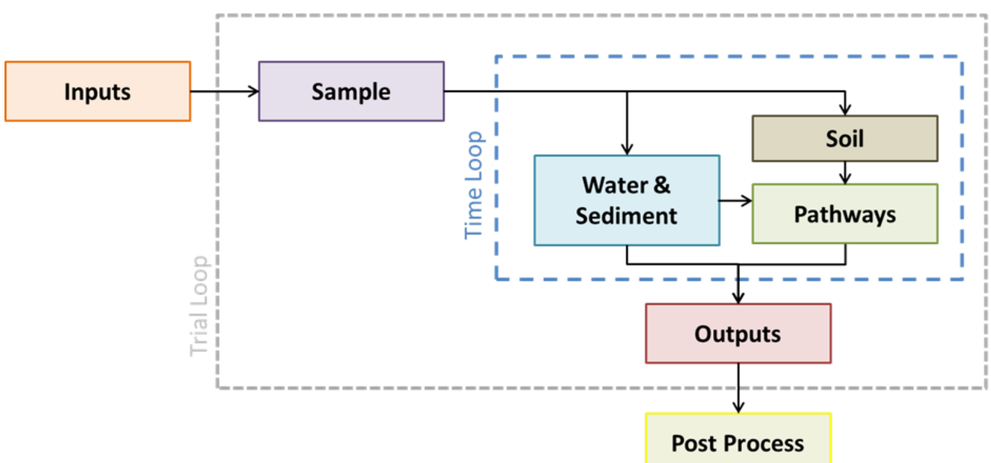


Figure 6. Schematic Overview of ADEPT

Inputs to ADEPT for the prediction of water and sediment quality include:

1. Treated mine water from the Mine Water Treatment Plant (MWTP) to Seru Bay
2. Groundwater loads to Cigar Lake, Longyear Bay, Aline Lake, and Aline Creek

For this application, the model was run using annual time steps to assess the variations in watershed flows on receiving water quality. Data on MWTP flow and concentrations from 2013 to 2020 were used to show the influence of past operations on water and sediment concentrations, as well as for calibration of the model to make sure ADEPT provides reasonable estimates of concentrations. Treated mine water source terms were developed for future assumptions used to model three additional phases of the Operation including Remainder of Cigar Lake Life of Mine (2021-2036), Decommissioning (2037-2041), and Post-decommissioning (2042 onwards). The present modelling is consistent with the modelling carried out for the 2017 ERA with the following exceptions:

1. The model in the 2017 ERA used monthly time steps and the present model considered annual time steps.
2. Cigar Lake and Aline Lake were considered in the present model in addition to the waterbodies modelled in the 2017 ERA (Seru Bay, Longyear Bay, and Seru Bay Junction)
3. Additional data collected since the 2017 ERA was incorporated into the present model.

Groundwater loads were developed by Golder Associates (Golder) to estimate loads from current and expected surface sources at the site. The results from the groundwater modelling were used in ADEPT as loads to the receiving waterbodies and the subsequent transfer through the environment. The results from air dispersion modelling, which was completed separately following the Saskatchewan Air Quality Modelling Guideline (ENV 2012), were used in ADEPT to estimate changes in the environment in things like soil, lichen, and plants. Concentrations in food items (e.g., fish, berries, moose) are estimated in ADEPT using transfer factors. Transfer factors relate the concentration in one part of the environment to another, for example between water and fish. These transfer factors are based on a large database of information that CanNorth has gathered over time from northern Saskatchewan. Literature values are used for the intake to flesh concentration for game (e.g., Baes et al. 1984; IAEA 2010; CSA 2008), although adjustments are made to make sure the concentrations are close to measured data in wild game. The ADEPT model is used to predict concentrations in the future and these predictions are based on an understanding of environmental processes and potential future conditions. Measured concentrations of COPCs were used in the assessment from many years of environmental monitoring programs. These data were used to calibrate and confirm that the ADEPT model used in the assessment provides a reasonable representation of the site environment. Estimates of future concentrations were made based on expected operational activities, source loads, precipitation, etc. As stated, these are the best estimates, but it is still a theoretical exercise. Distributions were assigned based on data to predict a range for potential future observations.

Uncertainty

There is uncertainty in the exposure assessment, particularly with environmental modelling and intake rates. For the modelling, the measured COPCs concentrations used in the assessment were based on data from many years of environmental monitoring programs. These data were used to calibrate and confirm that the ADEPT model used in the assessment provides a reasonable representation of the site environment. Distributions were assigned based on data to predict a range for potential future observations. The modelling of future conditions has been completed using reasonable estimates, but it remains only an illustration of what might happen. Intake rates of country foods were based on a survey of local communities, but distributions were applied to human exposure assumptions, in order to capture variability between individuals and acknowledge the uncertainty in the assumptions.

HHRA Toxicity Assessment

Toxicity is the potential of a COPC to cause some type of adverse effect. The toxicity depends on the amount of the COPC taken into the body (generally termed the intake or dose) and the length of time a person is exposed. Every COPC has a specific dose and duration of exposure that is necessary to produce a toxic response in humans. Toxicity assessments generally involve the evaluation of scientific studies, based either on laboratory animal tests or on workplace exposure investigations, by a number of experienced scientists in a wide range of scientific disciplines in order to determine the maximum dose that a human can be exposed to without having an adverse health outcome. However, it should be noted that exposure above this level does not mean that an effect will occur but instead means that there is an increased risk of an effect occurring that should be investigated further to determine.

The toxicity reference values (TRVs) used in the assessment for both carcinogenic and non-carcinogenic endpoints were from governmental agencies such as Health Canada (2010c, 2021), U.S. EPA (2020), Environment Canada/Health Canada (2011), ATSDR (2004, 2013), and JECFA (FAO and WHO 2011).

Exposure to gaseous pollutants (NO_2 and SO_2) and dust are assessed using health-based air quality guidelines. The World Health Organization provides the most current health-based values available (WHO 2021), and they were used as health-based values in the assessment, where available.

Assessment of radiation exposures to members of the public is based on estimation of the incremental dose of the project or site. Potential effects from radiation were assessed against an incremental dose limit of 1000 μSv/y (1 mSv/y) recommended by the CNSC for the protection of members of the public and dose constraint of 300 μSv/y (0.3 mSv/y) recommended by Health Canada in the Canadian NORM Guidelines (Health Canada 2000). Doses below this level are considered as "unrestricted," and no further action is needed to control doses or materials.

Uncertainty

It is acknowledged that there is uncertainty associated with the use of the TRVs selected for the assessment. It is currently not possible or practical to develop approaches to evaluate the validity of the toxicity benchmark assumptions in the overall assessment. As improvements occur in toxicological research and assessments, the uncertainties may be reduced.

Another area of uncertainty in the risk assessment is the impact of multiple COPCs. When dealing with toxic constituents, there is a potential interaction with other chemicals that may be found at the same location. It is well established that synergism, potentiation, antagonism, or additivity of toxic effects occurs in the environment. A quantitative assessment of these interactions would be constrained, as there is not an adequate basis of toxicological evidence to quantify these interactions. This results in uncertainty related to COPCs interactions and associated risks.

HHRA Risk Characterization

In the risk characterization phase, the information from the exposure assessment and toxicity assessment are combined to determine the potential for an adverse outcome based on the environmental conditions and use of the area.

The HHRA was completed using a probabilistic assessment in order to capture a likely estimate (average) and a reasonable range (5th percentile, 95th percentile) of potential future exposures. The HHRA considered:

- Radionuclides
  - It is important to remember that people are exposed to radiation from natural sources all the time. This includes radiation from cosmic rays; radionuclides in air, water, and food; radon; and gamma radiation from radioactive materials in the soil, rocks, and building materials used in homes. Worldwide, the normal range of average exposures to natural background radiation has been reported as about

1,000 to 13,000 μSv/year (UNSCEAR 2008b, Annex B). The expected dose (above background) for the worker and lodge resident are well below the CNSC for the protection of members of the public and the dose constraint recommended by Health Canada.

- Non-carcinogenic COPC
  - The ADEPT model included pathways such as drinking water, soil contact, and food obtained from local sources (meat, fish, berries, and other traditional food consumption). Not all food sources were included; therefore, typical intakes for general Canadians (Health Canada 1994) from store-bought market foods (e.g., meat, dairy, grains, and vegetables) were also considered in the overall assessment. In addition, background exposure to water and soil, while not in the area, was also included. The predicted daily intakes for the non-carcinogenic, non-radionuclide COPCs (i.e., arsenic, cobalt, copper, lead, molybdenum, nickel, selenium, uranium, and zinc) were compared to the TRV. Note that arsenic is included for the assessment of the non-carcinogenic and carcinogenic endpoints. With the exception of arsenic, nickel, and zinc, all intakes are well below the TRVs. For arsenic, nickel, and zinc the exposure is primarily from the general Canadian intakes (i.e., supermarket food such as milk and cereal) and the contribution from the Operation is negligible. In addition, no concerns are expected based on predicted air concentrations for non-carcinogenic, non-radionuclide COPCs.

- Carcinogenic COPC
  - Arsenic is considered to be a carcinogen via the oral and inhalation exposure routes and nickel is carcinogenic through the inhalation pathway. Incremental cancer risks resulting from exposure from the Operation were calculated for the worker adult receptors and a lifetime for the Waterbury Lodge operator. The incremental cancer risks are well below the negligible cancer risk level of 1 per 100,000 at both the average and 95^{th} percentile prediction levels.

- Gaseous pollutants and particulate
  - The concentrations of NO_2 and SO_2 are below the health-based benchmarks. Results generated using conservative air emissions scenarios show that, while the PM_{10} concentrations and $PM_{2.5}$ (with background) are predicted to exceed the health-based criteria at the Cigar Lake Camp, the maximum predicted concentration of PM_{10} is around 50 times lower than the occupational exposure limit. There are no exceedances expected for the Waterbury Lodge Resident. The

background concentrations are conservative and account for 50% of the criteria on their own. Measurements from hi-vol monitoring stations are collected regularly at the Operation and indicate only occasional exceedances of the 24-hour standard for particulate matter, which were attributable to smoke from forest fires in the area, high winds and blowing sand, and maintenance activities on the road near the sampling station. Therefore, conservative background concentrations and emissions estimates used in the air modelling assessment, along with the assumption of the worst-case conditions occurring simultaneously with unfavourable meteorological conditions, contribute to an overestimate of the predicted particulate matter exceedances.

While evaluation of airborne COPCs show there is the potential for particulate at the Cigar Lake Camp to exceed benchmarks under conservative operational scenarios, further review indicates that the likelihood of negative outcomes at the Cigar Lake Camp is considered to be low. The results for the Upper-bound scenario are consistent, there are no potential concerns indicated for human receptor exposures to radionuclides, non-radionuclides, and carcinogenic non-radionuclides COPCs.

Cameco is a founding partner in the Eastern Athabasca Regional Monitoring Program (EARMP), which conducts independent regional monitoring downstream of uranium operations in northern Saskatchewan, including Cigar Lake, to ensure that there are no cumulative influences on the regional watershed. The 2021 10-year summary of the EARMP highlighted that the monitoring of streams and lakes downstream of uranium operations in northern Saskatchewan shows that the receiving environment remains protected. Based on air monitoring results, there is no influence on regional air quality.

The Community Based Environmental Monitoring Program (CBEMP) is also in place for the Athabasca Basin and the monitoring program regularly demonstrates that country food samples collected from the communities, including Hatchet Lake First Nation and Wollaston Lake, were not negatively influenced by active uranium operations in the region and the foods remain safe for consumption. In 2023, a HHRA was completed based on information collected through the CBEMP, using specific dietary rates and country food information.

The HHRA results show that community members should continue regularly eating locally harvested fish, wild game, berries and plants.

Uncertainty

There are many areas of uncertainty in a risk assessment. This is due to the fact that assumptions have to be made throughout the assessment either due to data gaps, to environmental fate complexities, or in the generalization of receptor characteristics. In recognition of these uncertainties, some conservative assumptions are used throughout the assessment to ensure that the potential for a negative effect would not be underestimated.

Although there are many uncertainties inherent in a risk assessment (receptor characterization, exposure, and TRVs) and in the prediction of future concentrations (effluent assumptions, precipitation, and transfer factors), the assessment was completed with reasonably conservative assumptions and within a probabilistic framework in order to capture a likely estimate (average) and a reasonable range (5th percentile, 95th percentile) of potential future exposures. The reliance on data collected through environmental monitoring programs reduces the uncertainties in the assessment.

HHRA Conclusions

The HHRA concluded that there are no expected risks to human health from radionuclides, non-carcinogenic COPCs, or carcinogenic COPCs related to the Cigar Lake Operation. Human health in the vicinity of the Operation remains protected.

Ecological Risk Assessment

EcoRA Problem Formulation

A number of ecological receptors were selected to represent the range of biota expected to frequent the area around the Operation. This includes aquatic biota (e.g., benthic invertebrates, fish), terrestrial plants (e.g., foliage, lichen), semi-aquatic animals (e.g., waterfowl, muskrat, beaver), and terrestrial animals (e.g. hare, blackbird, fox, caribou).

Ecological receptors were selected based on biological surveys completed in the vicinity of the Operation as well as other considerations including stakeholder interviews and input from local communities. Presence or absence of species at risk (SAR), either provincially or federally, were also identified.

Cameco regularly conducts a review of wildlife species at risk that may be found in areas they operate, including the Cigar Lake Operation area. Review of SAR included a *Species at Risk Act* (SARA) public registry database search in December 2020 (Government of Canada 2020). The data were screened for species present in Saskatchewan with a status of ‘Threatened’, ‘Endangered’ or ‘Special Concern’, then further investigated to determine if there may be present in the vicinity of the Operation. Field surveys and site observations then informed which species were selected as SAR at the Operation. Five birds (rusty blackbird, common nighthawk, barn swallow, bank swallow, and olive-sided flycatcher) and three mammals (woodland caribou, little brown bat, and wolverine) were identified as SAR potentially present in the site area and have been observed at the Cigar Lake site. These species were selected for inclusion in the current assessment.

Aquatic Receptors

The aquatic species selected for this assessment cover all trophic levels found in watershed systems including:

- **Primary producers** – are the lowest level in the aquatic food chain. These are organisms which use inorganic molecules and the sun to produce food. Examples of primary producers in the aquatic environment are phytoplankton and other aquatic plants.

- **Primary consumers** –feed on the primary producers. Members of this group include zooplankton, benthic invertebrates and small fish. The larvae of some benthic invertebrates are an important food source for benthic and juvenile pelagic fish, while the adult stages of some species are capable of flight and are consumed by birds and bats, creating a link between the aquatic and terrestrial ecosystems.

- **Secondary consumers** - include fish whose diet consists largely of benthic invertebrates and smaller fish. Benthic fish species which were observed in Seru Bay include white sucker, lake whitefish, longnose sucker, lake chub, burbot, and slimy sculpin.

- **Tertiary consumers** - are the top of the aquatic food chain and include species of predatory fish. Predatory fish species noted in the Cigar Lake area include northern pike and lake trout in Seru Bay.

- **Amphibians** –The boreal chorus frog and the wood frog were observed during surveys in the area and were, therefore, considered in the assessment.

No reptiles were identified through the SARA database screen or through survey of the Cigar Lake area and, therefore, were not considered.

Terrestrial Species

The terrestrial species selected for this assessment include:

- Terrestrial Plants - The Cigar Lake area is dominated by jack pine communities. Potential influences on terrestrial plants were assessed by consideration of foliage, woody vegetation, fruits and flowers, roots, and lichen.

- Terrestrial Invertebrates - Terrestrial invertebrates were considered in the assessment. Terrestrial invertebrates include soil-dwelling organisms, such as earthworms as well as terrestrial-based insects (e.g., ants, spiders, bees).

Wildlife

The wildlife species rely on both the aquatic and terrestrial environments and include:

- Aquatic Birds – these migratory birds rely primarily on the aquatic environment

  - *Mallard* –The mallard is a dabbling duck observed in Seru Bay and across the broader Cigar Lake Operation study area.

  - *Common Merganser* –The merganser is a diving duck whose consumes fish. Common mergansers were observed in Seru Bay.

  - *Lesser Scaup* –The scaup is a diving duck that have been detected across the broader Cigar Lake Operation study.

- Semi-Aquatic Mammals

  - *Beaver* – Beaver are found throughout most of North America. They make their homes in lodges built either in the middle of a lake or pond or on the bank of a waterway. Surveys of Seru Bay found limited beaver activity in the study area with only inactive or old houses present. Interviews with local trappers found that while trapping in the Cigar Lake area is primarily done for furs, they will eat beaver.

  - *Muskrat* – Muskrat make their home in permanent wetland areas which contain cattails, bulrushes and/or sedges. Surveys found evidence that muskrat frequent the Seru Bay area with muskrat scat and feeding areas noted.

- *Mink* – Mink are carnivorous with the largest portion of the diet made up of fish. Mink scat and tracks were widely detected in the Cigar Lake Operation area.

- Terrestrial Birds
  - *Willow Ptarmigan* – Evidence of ptarmigan/grouse species (tracks and droppings) were observed in the Cigar Lake Operation area during terrestrial survey work.
  - *Bald Eagle* – Bald eagles are carnivorous in nature, eating primarily fish. Bald eagles have been observed during surveys.
  - *Rusty Blackbird* – The rusty blackbird is listed as special concern in the SARA registry. Rusty blackbirds were observed in the Cigar Lake Operation study area and serves as a surrogate species for other SAR species such as the common nighthawk, barn swallow, bank swallow, and olive-sided flycatcher.

- Terrestrial Mammals
  - *Masked Shrew* – The shrew is an insectivore and is present in the Cigar Lake study area.
  - *Snowshoe Hare* – The hare is an herbivore. Winter tracking field programs found abundant snowshoe hare tracks throughout the study area.
  - *Little Brown Bat* – The little brown bat is listed as endangered in the SARA registry. Little brown bats were observed roosting in the Waterbury Lake area during recent site surveys.
  - *Moose* – Moose can be found in both forest and wetland, especially areas with diverse browse species. An aerial ungulate survey identified two moose in the greater Cigar Lake site area. In addition, moose tracks and droppings have been observed within site area.
  - *Black Bear* – The black bear prefers habitat of deciduous or coniferous forests, swamps, and berry patches. Black bear scat was noted during terrestrial survey activities.
  - *Red Fox* – Fox tracks and droppings were noted during terrestrial survey work.

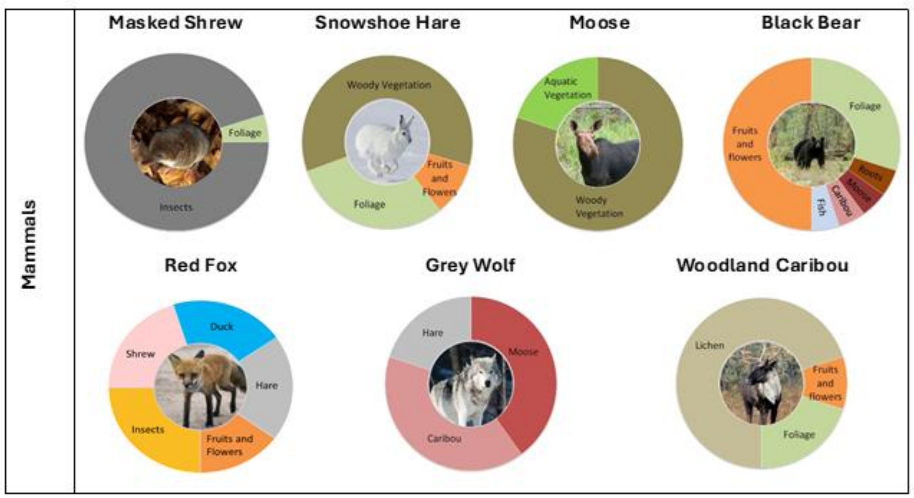
- *Grey Wolf* – The grey wolf is a carnivore with a large reported foraging range. Wolf tracks and scat were noted during terrestrial survey activities and have been detected by trail. The wolf is also a surrogate for wolverine.

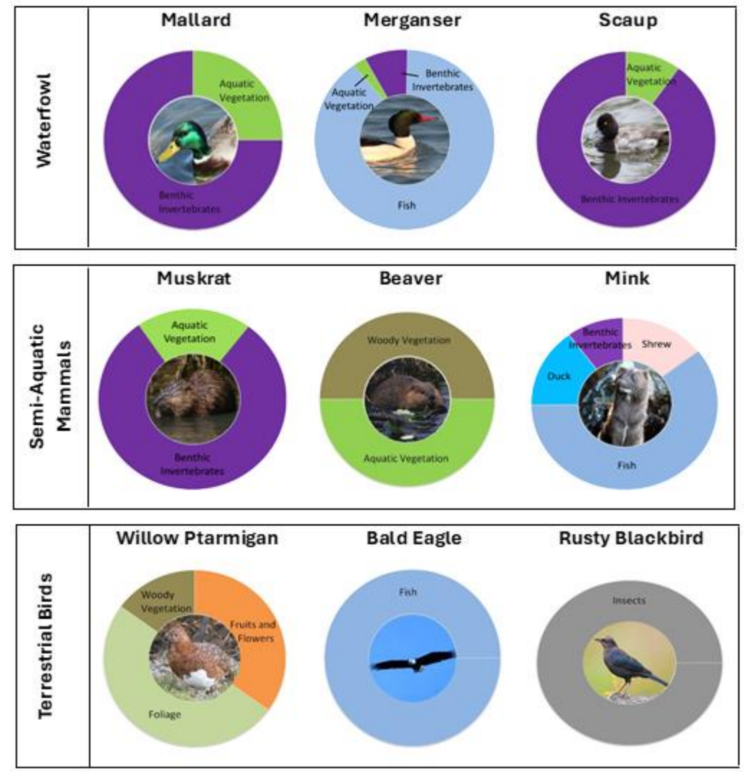
- *Woodland Caribou* – Woodland caribou are listed as threatened on the SARA registry and their home range includes northern Saskatchewan. They are present in the area of the Cigar Lake Operation

Species that are linked strongly to the aquatic environment through their preferred habitat and dietary intakes are evaluated on a waterbody basis, while those that consume more terrestrial-based foods (e.g., ptarmigan and woodland caribou) are evaluated on an airshed basis.

An overview of the characteristics of the selected mammals and birds is provided in Table 1.

Table 1. Summary of wildlife receptors and their diet





Assessment endpoints are set for the feature and functions of the environment that are to be considered in the evaluation. These endpoints are typically broad and represent the key attributes of the ecosystem or species that are important for maintaining ecological integrity and function. The assessment endpoints include:

- Aquatic receptors: healthy, sustainable populations.
- Terrestrial receptors: healthy, sustainable populations.
    - SAR were assessed on an individual level.

Measurement endpoints are quantifiable parameters that are used to assess the assessment endpoints. A measurement endpoint is defined as "a quantitative summary of the results of a toxicity test, a biological study, or other activity intended to reveal the effects of a substance" (Suter 1993). For this assessment, measurement endpoints are generally toxicity values, discussed later in the summary.

Consistent with N288.6-12, the receptor exposure pathways for the ecological assessment are shown in Table 2.

Table 2. Ecological Exposure Pathways

Receptor Group	Exposure Pathways			
	Soil	Surface Water	Sediment	Food
Terrestrial invertebrates	✓	NR	NR	NR*
Terrestrial plants	✓[a]	NR	NR	NR
Aquatic birds	NR	✓	✓	✓
Terrestrial birds	✓	✓	NR	✓
Semi-Aquatic mammals	NR	✓	✓	✓
Terrestrial mammals	✓	✓	NR	✓
Amphibians[b]	NR	✓	✓	NR*
Fish	NR	✓	✓	NR*
Aquatic plants	NR	✓	✓	NR
Aquatic invertebrates	NR	✓	✓	NR*

Note: VCs were not identified for reptiles; NR – not relevant; ✓ - assessed; a - exposure to air as well as soil; b - may be assessed using fish as surrogate. * Food chain modelling not conducted for these receptors; food consumption included in the effects assessment on soil or water.

A CSM is a representation of the biological, physical and chemical processes that determine the ways that constituents move from sources through the environmental media to environmental receptors. The ecological CSM for the Operation was shown in Figure 5.

EcoRA Exposure Assessment

The ADEPT model was used to estimate the concentration in the environment, including future releases from the Cigar Lake Operation. This model was described in the previous section.

For smaller receptors that get the majority of their dietary intakes from the aquatic environment (i.e., waterfowl, muskrat, beaver, and mink), each species was evaluated on individual lake segments (e.g., Seru Bay South). For migratory species such as ducks, which establish nests on specific waterbodies while in the study area, all of their exposure is obtained from the specified waterbodies. Although these species are known to not be present at the site year-round, they are present for a sufficient duration during a sensitive life stage that no averaging was used in the assessment. Larger receptors with a close connection to the aquatic environment were assessed at Waterbury Lake. It has been assumed that larger ecological receptors such as the moose roam around Waterbury Lake. Small terrestrial-based receptors, such as the shrew and hare were evaluated based on a 1-km^2 airshed around the Cigar Lake Operation; the blackbird, ptarmigan, bear, and fox were assumed to roam around a 5-km^2 airshed around the Cigar Lake Operation; and larger receptors such as wolf and woodland caribou were evaluated on a 100-km^2 airshed.The transfer of COPCs to environmental components of the aquatic and terrestrial environments is completed using the equations provided in the following sections. Transfer factors (TFs) were used in the pathways assessment to evaluate the transfer of COPCs through the various environmental media considered in the assessment. Transfer factors relate the concentration in Where possible, data collected through the monitoring programs were used to develop site-specific transfer factors (i.e., fish) or validate calculated concentrations (i.e., aquatic vegetation, benthic invertebrates, soil, and vegetation). In some cases, literature values were used in the absence of site data. There is some uncertainty involved in the use of these transfer factors to predict concentrations in environmental media.

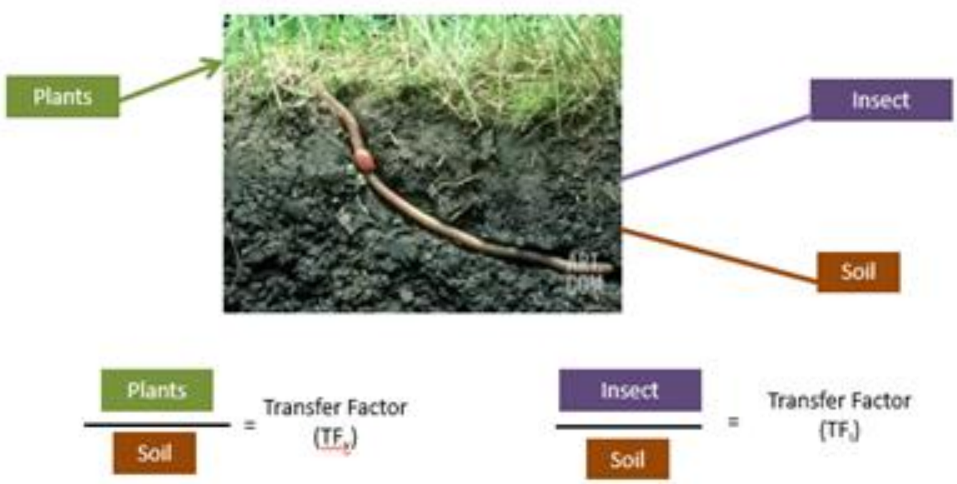


Figure 7. Example of transfer factors

AQUATIC ENVIRONMENT

ADEPT calculates the concentrations in components of the aquatic environment, such as aquatic vegetation, benthic invertebrates, fish, phytoplankton, and zooplankton, using COPC- and receptor-specific transfer factors. Equation (8) provides the calculation for aquatic vegetation, fish, benthic invertebrates, phytoplankton, and zooplankton.

$$C_t = C_w \times TF_{w-t}$$

$$C_t = C_s \times TF_{s-t} \quad (8)$$

Where:

C_t	= Tissue COPC concentration in aquatic vegetation, fish, benthic invertebrates, phytoplankton, and zooplankton (g/g (ww) or Bq/g (ww))
C_w	= COPC concentration in water (g/m^3 or Bq/m^3)
C_s	= COPC concentration in sediment (g/g (dw) or Bq/g (dw))
TF_{w-t}	= COPC transfer factor for water-to-tissue for aquatic vegetation, fish, phytoplankton, and zooplankton (m^3/g (ww))
TF_{s-t}	= COPC transfer factor for sediment-to-tissue for benthic invertebrates (g (dw)/g (ww))

Predicted concentrations in fish are calculated using a "hockey stick" model (Brix et al. 2005; Toll Environmental 2005), where minimum COPC concentrations in fish are defined for low concentration environment and a transfer factor is used to calculate fish concentrations in higher concentration environments. Therefore, the fish concentration never drops below the minimum concentration.

A different approach was taken for selenium due to its bioaccumulation in the aquatic environment. Presser and Luoma (2010) provide support for a trophic transfer and bioaccumulation model for fish for selenium, specifically. Following their work, ADEPT considers the transfer of non-radionuclide COPCs in the environment to fish as:

Water -> Particulate (sediment) -> Invertebrates (benthos) -> Fish

TERRESTRIAL ENVIRONMENT

Vegetation

In the terrestrial environment, concentrations in vegetation are calculated using transfer factors from soil to various types of functional groups, based on plant parts and mode of uptake from the environment. The transfer of COPCs from soil to plant is based on N288.1-20 (CSA 2020), with consideration of derived transfer factors to represent composite uptake and translocation processes that occur. The transfer from soil to plant occurs for all vegetation types, except for lichen. Transfer factors for the soil-to-foliage, -woody veg, and -fruits and flowers were developed from data collected from sites in northern Saskatchewan. Data were not available for the development of transfer factors for soil-to-roots based on field data. Therefore, literature values were used for roots. Lichen concentrations are not connected to soil; therefore, transfer factors for soil-to-lichen are not used in the model.

Insects

Two types of insects are considered within the model: emergent aquatic insects and terrestrial insects. Emergent aquatic insects are species such as caddisflies, mayflies, and damselflies that originate in the aquatic environment. Terrestrial insects are land-based species such as earthworms, spiders, ants, beetles, and bees.

Based on the available information, concentration factors were derived for aquatic emergent insects, based on benthic invertebrates. Concentrations in terrestrial insects were estimated from soil concentrations using transfer factors available in the ERICA database (Beresford et al. 2007) and IAEA (2014). Molybdenum was not included in these references, so data presented in Van Gestel et al. (2011)[1] for earthworms was used.

Wildlife Receptors

Ecological receptor intakes are calculated following equation (9).

$$D_{ing} = \frac{\sum_x C_x \times I_x}{W} \quad (9)$$

Where:

D_{ing} = COPC intake by eco receptor (g/kg-d or Bq/kg-d)

C_x = COPC concentration in ingested item x (g/m^3 or Bq/m^3, g/g (dw) or Bq/g (dw), g/g (ww) or Bq/g (ww))

I_x = Ingestion rate of item x (m^3/d, g (dw)/d, g (ww)/d)

W = Body weight of eco receptor (kg)

Concentrations in ecological receptors are calculated using the intake equations above and for the radiological dose assessment by applying a transfer factor for feed-to-flesh, as shown in equation (10).

$$C_t = D_{ing} \times W \times TF_{feed-t} \quad (10)$$

Where:

C_t	=	Tissue COPC concentration in eco receptor (g/g (ww) or Bq/g (ww))
D_{ing}	=	COPC intake by eco receptor (g/kg-d or Bq/kg-d)
W	=	Body weight of eco receptor (kg)
TF_{feed-t}	=	COPC transfer factor for feed-to-flesh of eco receptor i (d/g (ww))

Inhalation exposures are generally minor relative to soil and food ingestion exposures and can be ignored in most ecological risk assessments (CSA 2012). Similarly, although dermal exposure through direct contact is potentially a complete exposure pathway for birds and mammals, this exposure pathway is usually low due to fur and feathers, duration of exposure, and the low relative contribution compared with oral exposures (U.S. EPA 2003).

Radioactivity

The assessment of potential effects from exposure to radioactive constituents involves estimation of the combined (total) dose which a receptor may receive from radionuclides taken into the body as well as from exposure to radiation fields in the external environment. In addition, it is standard practice to take into account differences in alpha, beta, and gamma radiation. An RBE of 10 for internally deposited alpha radiation was used in this assessment, consistent with the N288.6-12 CSA Standard (CSA 2012). The Dose Coefficients (DCs) used for estimating dose in this assessment were obtained from Amiro (1997) and Blaylock et al. (1993).

Uncertainty

There are several areas of uncertainty in the Exposure Assessment including estimation of future water and sediment concentrations, biota concentrations, and COPC intake by receptors. Transfer factors were used in the pathways assessment to evaluate the transfer of COPCs through the various environmental media considered in the assessment. In some cases, literature values were used in the absence of site data. There is some uncertainty involved in the use of these transfer factors to predict concentrations in environmental media. Where possible, data collected through the monitoring programs were used to develop site-specific transfer factors (i.e., fish) or validate calculated concentrations (i.e., aquatic vegetation, benthic invertebrates, soil, and vegetation).

The characteristics (food, water, and soil consumption) of ecological receptors were obtained from literature. These values are generally obtained from animals in captivity and may not be fully representative of free-range animals in the wild; however, distributions are used in an attempt to capture variability and uncertainty.

EcoRA Effects Assessment

In the effects assessment, toxicological benchmarks or toxicity reference values (TRVs) are selected. These are values below which environmental effects are not expected. The values are selected from regulatory guidelines and scientific literature. An emphasis is placed on long-term exposure. These TRVs are used in risk assessments to judge whether the predicted (estimated) exposures (or doses or intakes) may potentially have a negative effect on ecological species. Site-specific information was incorporated into the selection process for TRVs where available.

Water Quality

The potential influence of the project on water quality in the receiving environment was evaluated by comparing predicted concentrations of COPCs in water to Water Quality Guidelines (WQGs) available at the time. Selected water quality guidelines include the Saskatchewan Environmental Quality Guideline (SEQG, GS 2019) for the protection of aquatic life; guidelines for COPCs without SEQG values were taken from other sources such as the Canadian Water Quality Guidelines (CWQG) for the Protection of Aquatic Life (CCME 2019), Health Canada Guidelines for Canadian Drinking Water Quality (Health Canada 2019), or Environment Canada Federal Environmental Quality Guidelines (FEQG, EC 2019). Available WQGs for the protection of aquatic life for the COPCs are summarized in Table 7.

Table 3. Water quality objectives and guidelines for the 2021 ERA

Note: SEQG – Saskatchewan Environmental Quality Guideline for the Protection of Aquatic Life (Saskatchewan Environment 2019), EC – Environment Canada Federal Environmental Quality Guideline (EC 2019), HC – Health Canada drinking water guideline (Health Canada 2019), CWQG – Canadian Water Quality Guidelines (CCME 2019).

a – SEQG adopted the CEQG for unionized ammonia; it is incorrectly reported as 0.019 mg/L (as N) (which is the CEQG for unionized ammonia as NH3).

b – When the water hardness is 0 to <17 mg/L the guideline is 0.04 μg/L; at hardness >280 mg/L is 0.37 μg/L.

c – When the water hardness is 0 to <82 mg/L the guideline is 2 μg/L; at hardness >180 mg/L is 4 μg/L.

d – When the water hardness is 0 to <60 mg/L the guideline is 1 μg/L; at hardness >180 mg/L is 7 μg/L.

e – When the water hardness is 0 to <60 mg/L the guideline is 25 μg/L; at hardness >180 mg/L is 150 μg/L.

COPC	WQG	Units	Note
Arsenic	5	μg/L	SEQG/CWQG Freshwater aquatic life (CCME)
Cobalt	0.78-1.8	μg/L	FEQG (EC), based on equation: $e^{[0.414 * \ln(hardness) - 1.887]}$
Copper	2-4	μg/L	SEQG: 0.2 $e^{[0.8545 * \ln(hardness) - 1.465]}$ (c)
Lead	1-7	μg/L	SEQG: $e^{[1.273 * \ln(hardness) - 4.705]}$ (d)
Molybdenum	31,000	μg/L	SEQG
Nickel	25-150	μg/L	SEQG: $e^{[0.76 * \ln(hardness) + 1.06]}$ (e)
Selenium	1	μg/L	SEQG
Uranium	15	μg/L	SEQG
Zinc	30	μg/L	SEQG
Thorium-230	0.6	Bq/L	Health Canada for drinking water, Appendix A (HC)
Radium-226	0.11	Bq/L	SEQG
Lead-210	0.2	Bq/L	Health Canada for drinking water (HC)
Polonium-210	0.1	Bq/L	Health Canada for drinking water, Appendix A (HC)
Ammonia (un-ionized)	16 (a)	μg/L-N	SEQG/CWQG Freshwater aquatic life (CCME)
Chloride	120	mg/L	SEQG/CWQG Freshwater aquatic life (CCME)
Nitrate (NO_3-N)	3	mg/L-N	SEQG
Sulphate	128-429	mg/L	SEQG: 128 mg/L where hardness is 0 - 30 mg/L; 218 mg/L where hardness is 30 – 75 mg/L; 309 mg/L where hardness is 75 – 180 mg/L; 429 where hardness is 180 – 250 mg/L
TDS	500	mg/L	Health Canada for drinking water (HC)

SEDIMENT QUALITY

Table 4 outlines sediment quality benchmarks used in the assessment. The values from Thompson et al. (2005) are specific to uranium-bearing regions of Canada (e.g. northern Saskatchewan and northern Ontario) and are considered CNSC working reference values. Benchmark values from Burnett-Seidel and Liber (2013) were determined specifically for northern Saskatchewan waterbodies and are considered in the Eastern Athabasca Regional Monitoring Program (EARMP), which was established to monitor long-term changes in the aquatic environment downstream of uranium mines in northern Saskatchewan.

Table 4. Sediment quality toxicity benchmarks

Note: Constituent concentrations/activity levels are based on sediment dry weight.

- - no benchmark available, LEL - lowest effect level, SEL - severe effect level, REF – reference level, NE2 – no effect level.

a – Values for copper and lead were derived by Burnett-Seidel and Liber (2013) however, the derived NE values did not change with the incorporation of additional effects criteria due to a lack of influence from the uranium operations on the concentrations of these metals in sediment and were not recommended for the evaluation of sediment at Saskatchewan uranium mining and milling operations.

b – No values were provided by Thompson *et al*. or Burnett-Seidel and Liber so zinc benchmarks are an Interim Sediment Quality Guideline (ISQG) and Probable Effect Level (PEL) from CCME (2017).

Constituent	Thompson et al. 2005		Burnett-Seidel and Liber 2013	
	LEL	SEL	REF	NE2
Metal (µg/g)				
Arsenic	9.8	346.4	20.8	522
Cobalt	-	-	-	-
Copper	22.2	268.8	-[a]	-[a]
Lead	36.7	412.4	-[a]	-[a]
Molybdenum	13.8	1238.5	23	245
Nickel	23.4	484	21.4	326
Selenium	1.9	16.1	4.0	29.7
Uranium	104.4	5874.1	96.7	2296
Zinc	123[b]	315[b]	-	-
Radionuclide (Bq/g)				
Radium-226	0.6	14.4	-	-
Thorium-230	-	-	-	-
Lead-210	0.9	20.8	-	-
Polonium-210	0.8	12.1	-	-

As seen from the table, Thompson et al. (2005) have proposed both Lowest Effect Level (LEL) and Severe Effect Level (SEL) for benchmarks. The LEL represents the COPC concentration below which harmful effects on benthic invertebrates are not expected. The SEL represents the COPC concentration above which harmful effects on benthic invertebrates are expected. Caution is to be employed when using the SEL values, as they may not be a reliable predictor of potential effects. Burnett-Seidel and Liber (2013) have established reference (REF) and No Effect (NE2) sediment values. REF benchmarks refer to locations upstream of mining or milling activities or within separate, but nearby drainages, while NE2 values refer to exposed areas with no significant effect on benthic invertebrate abundance, richness, and evenness. These benchmark values were derived using an approach based on no-effects field data that reflect local benthic invertebrate tolerances to elevated concentrations of some metals in sediment.

The Saskatchewan Ministry of Environment strongly recommends using the NE2 and REF guidelines as the primary criteria, as they are based on data collected from Northern

Saskatchewan ecoregions as per the recommendations by the Joint panel regarding the development of Regional Sediment and Water Guidelines for Northern Saskatchewan. Given that, the Burnett-Seidel and Liber (2013) sediment values are considered to be the most applicable benchmarks for use at sites in northern Saskatchewan, such as the Cigar Lake Operation

A weight-of-evidence approach where the use of these sediment quality benchmarks and actual data collected in the field is provided in this assessment.

Aquatic Toxicity Reference Values

The approach for assessing the potential effects on aquatic biota (aquatic plants, phytoplankton, zooplankton, benthic invertebrates and fish) exposed to non-radionuclide COPCs is through the development and application of species sensitivity distributions (SSDs). This method combines all relevant toxicity data to estimate the relationship between surface water concentrations and the toxicity of individual aquatic biota. The SSD approach has been adopted for the development of the site-specific water quality objectives and is therefore supported by the CCME protocol (CCME 2007). The SSD approach is consistent with N288.6-12 (CSA 2012) and has been used in previously approved assessments for the uranium industry.

Data used to inform the SSD for aquatic biota was primarily sourced from the U.S. EPA ECOTOXicology database (ECOTOX) and infilled with literature data where necessary. Data were then summarized and screened to meet the following criteria:

- freshwater tests;
- chronic;
- relevant species to Canada or relevant non-resident species;
- ECx, ICx, MATC, and LCx endpoints (where x >=10); and
- inorganic chemical form.

Where multiple values were available for a species, growth and reproductive endpoint were preferentially selected according to the following:

$$\begin{Bmatrix} EC_{10-25} \\ IC_{10-25} \\ MATC \end{Bmatrix} > NOEC > LOEC > \begin{Bmatrix} EC_{50} \\ IC_{50} \end{Bmatrix} > LC_x$$

Where multiple preferential values were available, a geometric mean was calculated and assumed to be representative of that species. The SSD Master V3 tool (CCME 2013) was preferentially used to develop the curves for the SSDs and the U.S. EPA (U.S. EPA 2010) SSD generator was used where necessary. A scoring system was then used to evaluate the strength of the data used to develop the SSD. Points or penalties were assigned to an SSD according to each of the following criteria:

- number of species included
- presence of amphibian data
- if minimum species requirements according to the CCME (2007) were met
- NOECs, EC50s, lethal concentrations (LCs), or data from unreliable studies were needed to meet minimum requirements
- fit in the lower part of the curve

Finally, each SSD was assigned a letter grade according to the number of points it received to allow for a level of confidence in the dataset while also acknowledging uncertainty and limitations inherent to the development of an SSD.

The SSD approach using water concentrations was not used for selenium as it bioaccumulates in fish tissues. A number of criteria and benchmarks are available from various jurisdictions, which provide selenium concentrations in fish tissue associated with toxic effects. These benchmarks are summarized and discussed in Table 5.

Table 5.Selected selenium toxicity reference values for fish

ENVIRONMENTAL COMPONENT	UNITS	BENCHMARK	REFERENCE
Water Quality	µg/L	1.5	U.S.EPA (2016) water quality criterion for the protection of aquatic life – lentic aquatic systems
Fish Tissue	µg/g (dw)	11.3	U.S. EPA (2016) fish flesh/muscle concentration

Terrestrial Wildlife Toxicity Reference Values

TRVs were developed for avian and mammalian terrestrial species, based on Lowest Observable Adverse Effects Level (LOAEL) and/or No Observable Adverse Effects Level (NOAEL) data available in the literature. TRVs based on LOAEL values were considered for protection of populations.

Data used to derive TRVs were preferentially obtained from the Ecological Soil Screening Level (Eco-SSL) documents prepared by the U.S. EPA. The Eco-SSL screening process for wildlife toxicity data included a review of primary data sources. Preference was given to growth and reproductive endpoints; however, a check of the selected LOAEL values was made against data available for mortality and survival endpoints, as available. It should be noted that the use of LOAELs is consistent with N288.6-12 (CSA 2012), which states that selected benchmarks should correspond to the lowest exposure levels (e.g., LOAELs) associated with adverse effects.

The results of toxicity tests are generally for a limited number of test species. The approach was to find toxicity data for species that most closely represent those species evaluated in a particular assessment, referred to as surrogates. In choosing an appropriate surrogate test species, the closest taxonomic group is preferred. In the event that there was no satisfactory match among the test species to use in the risk assessment, default avian and mammalian TRVs were derived by combining all the available toxicity data from various species for each COPC.

For COPCs without Eco-SSL data (i.e., uranium), consideration was given for mammalian toxicity to a number of data sources including information from Sample et al. (1996), reproductive effects in mice from Feugier et al. (2008), as well as data from Linares et al. (2005) for reproductive effects in rats and data from Llobet et al. (1991) for reproduction effects (significantly reduced pregnancy rate) in mice. Data on growth effects in rats and rabbits from older studies by Maynard (Maynard and Hodge 1949; Maynard et al. 1953) were also considered. For avian species data were only available from Sample et al. (1996).

For terrestrial SAR identified for the assessment, little brown bat, woodland caribou, rusty blackbird, and wolf, consistent with N288.6-12 (CSA 2012) guidance, SAR are assessed at the individual, rather than population level. Therefore, NOAEL TRVs were selected for the assessment of surrogates for SAR.

Radioactivity

UNSCEAR (2008; Annex E) conducted a review of the available data on effects to aquatic biota, both direct and indirect effects (such as population shifts). Data on the effects of exposure to ionizing radiation were reviewed with a significant amount of information developed from follow-up observations of non-human biota in the zone around the Chernobyl nuclear power plant. There is a considerable range of endpoints and corresponding effects levels presented in the literature. Overall, it was concluded that a chronic dose rate of less than 400 µGy/h (9.6 mGy/d) to any individual in aquatic populations would be unlikely to have any detrimental effect at the population level. CSA N288.6-12 recommends adopting the UNSCEAR (2008) recommendations with respect to radiological dose benchmarks, therefore a dose benchmark of 9.6 mGy/d was adopted for aquatic biota.

UNSCEAR (2008; Annex E) conducted a review of the available data on effects to terrestrial biota, both direct and indirect effects (such as population shifts). They concluded that reproductive changes are a more sensitive indicator of the effects of radiation exposure than mortality and that mammals are the most sensitive animal organisms. Data on the effects of exposure to ionizing radiation were reviewed with a significant amount of information developed from follow-up observations of non-human biota in the zone around the Chernobyl nuclear power plant. There is a considerable range of endpoints and corresponding effects levels presented in the literature. Overall, UNSCEAR concluded that a chronic dose rate of less than 100 µGy/h (2.4 mGy/d) to the most highly exposed individuals would be unlikely to have significant effects on most terrestrial communities. CSA N288.6-12 recommends adopting the UNSCEAR (2008) recommendations with respect to radiological dose benchmarks, therefore a dose benchmark of 2.4 mGy/d was adopted for terrestrial biota (birds and mammals).

Uncertainty

It is acknowledged that there is uncertainty associated with the use of the TRVs selected for this assessment. However, for aquatic species, the use of the SSD curve in the assessment to evaluate the aquatic community as a whole adds to the robustness of the overall assessment. Given that no adequate toxicological database is available that determines the concentrations of COPCs that cause adverse effects in terrestrial ecological receptors, toxicity data from laboratory species such as rats and mice were used. Additionally, for terrestrial mammals and birds, toxicity information for a COPC was used regardless of its form in the test procedure, even though this may not be the same form used in the assessment (e.g., an oxide form compared to a more soluble form). It is hard to determine the effects of these assumptions.

Another area of uncertainty in the risk assessment is the effect of multiple COPCs. When dealing with toxic chemicals, there is a potential interaction with other chemicals that may be found at the same location. It is well established that synergism, potentiation, antagonism, or additivity of toxic effects occurs in the environment. A quantitative assessment of these interactions would be constrained, as there is not an adequate basis of toxicological evidence to quantify these interactions. This results in uncertainty related to COPCs interactions and associated risks.

EcoRA Risk Characterization

The approach taken in the ecological risk assessment component involved comparing estimation of exposures or doses for each of the selected receptors to toxicity benchmarks or TRVs. This comparison was undertaken by the calculation of Screening Index (SI) values, which provide an integrated description of the potential hazard, the exposure (or dose) response relationship and the exposure evaluation (AIHC 1992; U.S. EPA 1992). The index was calculated as shown in equation (11).

$$\text{Screening Index} = \frac{\text{Intake}}{\text{Toxicity Benchmark}} \tag{11}$$

For radionuclides, the total dose rate received by an ecological receptor was divided by the selected dose rate guideline to calculate a screening index value, as shown in equation (12).

$$\text{Screening Index} = \frac{\text{Dose Rate}}{\text{Dose Rate Guideline}} \tag{12}$$

The screening index values reported are not estimates of the probability of ecological impact. Rather, the index values are positively correlated with the potential of an impact, i.e. higher index values imply greater potential of an impact. Different magnitudes of the screening index have been used in other studies to screen for the potential ecological effects. In this study, an index value of 1.0 was used to examine the potential negative effects of COPC for aquatic receptors and for terrestrial receptors because background levels are incorporated within the calculations.

Temporal variations were addressed in the study by considering several time steps that span the period during which constituent concentrations are expected to change and then stabilize. Spatial variations were addressed by considering different locations within a watershed as previously outlined.

- Aquatic Biota:
  - Similar to the approved 2011 EIS, slight changes are expected to the concentrations within Seru Bay with only minimal change to the water quality in the rest of Waterbury Lake during the operational and decommissioning period. All COPCs are below the water quality guideline values in the modelled segments during operation of the Cigar Lake mine. In the post-decommissioning period, long-term groundwater loads from surface sources at the Cigar Lake Operation are expected to have minimal influence on the receiving environment; only cobalt in Aline Lake may exceed the water quality guideline. However, this is minor and the assessment is conservative; therefore, no negative influences on the aquatic communities of Aline Lake from groundwater loads in the post-decommissioning period are expected.
  - There are no exceedances of the selected sediment quality benchmarks in the operational, decommissioning, or long-term post-decommissioning period.
- Terrestrial Biota: The assessment of the potential influence of air emissions on plants and terrestrial invertebrates was done through the comparison of the soil concentration to the regulatory guideline. Potential negative influences on soil quality from the Cigar Lake Operation emissions to air are not indicated and therefore there are no concerns for terrestrial plants and terrestrial invertebrates from COPCs in soil.
- Terrestrial Biota: All SI values for wildlife are low, indicating no exceedances of the selected TRV. This includes SI values for species at risk.
- Radiological doses to aquatic and terrestrial biota are all low, indicating no potential concern. Consideration is also given to environmental monitoring from the site. Benthic invertebrate communities have been assessed in Seru Bay since 2010 (2010, 2013, 2016, and 2019). The benthic invertebrate monitoring program in Seru Bay and reference areas has remained consistent over time. Although there have been some differences observed between exposure and reference aquatic biota, there is no evidence at this time that the differences are due to mine-related exposure effects but reflect natural variations.

Supporting data suggest that the identified differences may be due to habitat differences between the exposure and reference areas, naturally low dissolved oxygen concentrations, and natural variation rather than mine-related exposure effects.

The ERA was completed using a probabilistic assessment in order to capture a likely estimate (average) and a reasonable range (5^{th} percentile, 95^{th} percentile) of potential future exposures. While long-term groundwater loads indicated a potential concern from cobalt exposure to aquatic biota in Aline Lake, this is minor and due to the conservative nature of the assessment, no negative effects are expected. The results for the Upper-bound scenario are similar. Although there are slightly higher zinc concentrations in sediment and selenium exposures to wildlife, no negative impacts are expected.

Uncertainty

There are many areas of uncertainty in a risk assessment. This is due to the fact that assumptions have to be made throughout the assessment either due to data gaps, to environmental fate complexities, or in the generalization of receptor characteristics. In recognition of these uncertainties, some conservative assumptions are used throughout the assessment to ensure that the potential for a negative effect would not be underestimated.

The assessment was completed with reasonably conservative assumptions and within a probabilistic framework in order to capture a likely estimate (average) and a reasonable range (5^{th} percentile, 95^{th} percentile) of potential future exposures. The reliance on data collected through environmental monitoring programs reduces the uncertainties in the assessment.

EcoRA Conclusions

The ERA concluded that there are no expected risks to ecological receptors from radionuclides or other COPCs related to the Cigar Lake Operation under the evaluated effluent scenarios. Consistent with previous assessments, the ERA demonstrated that the environment in the vicinity of the Cigar Lake Operation remains protected.

Climate Change

The Government of Canada published a report on Canada's Changing Climate in 2019 summarizing the state-of-the knowledge of the effects of climate change in Canada (Bush and Lemmen 2019). It reported that global warming and the impact of widespread warming are already evident in many parts of Canada and are projected to intensify in the future. The effects of climate change in Canada include more extreme heat (more frequent and more intense), less extreme cold, longer growing seasons, shorter snow and ice cover seasons, earlier spring peak streamflow, and changes to precipitation and wind gusts.

Climate change is complex, with many interactions and influences on various components of an ERA. The fate and transport of COPCs will be affected by climate change; however, there is uncertainty in how the many aspects of climate change will interact or combine effects within an exposure and risk assessment. There are expected to be influences on the mobility and bioaccumulation of the COPCs (i.e., constituent transport), as well as effects of climate change on the dose-response or the threshold dose (i.e., toxicity). There will also be physical changes in the environment, in terms of species presence and direct habitat effects, which will influence human interactions with the environment (i.e., exposure). There is a lack of information on the interactions or combined effects of the many climate change-associated events and processes.

The current assessment was completed in a probabilistic framework to capture some of the uncertainties associated with model assumptions and the probabilistic results account for, to an extent, possible future variations under a changed climate. This ERA was completed pursuant to CSA N288.6-12 and considered climate change in accordance with the standard.

ERA Conclusion

The results of the ERA for the receiving environment are summarized in Table 6 for the expected future releases. As expected, with the release of treated effluent to Seru Bay, the concentrations of COPCs are predicted to increase; however, only slight changes to the concentrations are expected within Seru Bay and only minimal to no change to the water quality outside of Seru Bay.

Table 6. Summary of Results to Receiving Environment

Sediment	Surface Water	Air
Predicted concentrations for COPCs are expected to remain below the selected sediment benchmarks.	Surface water levels are predicted to remain below all surface water quality guidelines, with the exception of cobalt in Aline Lake in the long-term due to groundwater loads.	There was no predicted influence on air quality from the Cigar Lake Operation.

Table 7 summarizes the results for the ecological and human health receptors for the expected future releases.

Table 7. Summary of Results to Receptors

Stressor Type	Human Receptors	Aquatic Biota	Terrestrial Biota
Scenario	The HHRA evaluated a Cigar Operation camp worker and a hypothetical Waterbury Lake Lodge operator	Assessment for a range of aquatic biota from benthic invertebrates (insects in the sediment at the bottom of the lake) to fish.	Assessment for terrestrial plants and wildlife. Selected species at risk (i.e., woodland caribou and blackbird) are protected on an individual basis (versus population basis).
Radiological	No expected risks to human health from radioactivity related to the Cigar Lake Operation.	No potential influence on aquatic biota are anticipated.	No potential influence on terrestrial biota are anticipated.
Non-Radiological	No expected risks to human health from COPCs released from the Cigar Lake Operation.	No potential influence on populations of aquatic biota are anticipated.	No potential concerns identified for terrestrial vegetation. No potential influence on terrestrial biota are anticipated from exposure to non-radionuclides COPCs.

The ERA meets the requirements of CSA N288.6-12. The results of the 2021 assessment are consistent with the findings from the 2011 EIS and previously accepted ERAs in that there are no significant risks posed to aquatic, terrestrial, or human receptors situated in the area surrounding the Operation. As such, it can be concluded that the environment and human health in the vicinity of the Cigar Lake Operation will remain protected.

Summary

The 2021 ERA was completed in accordance with CSA N288.6-12, and consistent with previously conducted environmental assessments and accepted ERAs, demonstrated that human health and the environment in the vicinity of the Operation remain protected.

This conclusion is consistent with the conclusions of the environmental impact statements and risk assessments that describe the site licensing basis for the Cigar Lake Operation.

References

A list of the documents that were used in the preparation of the ERA are provided below.

Acton, D.F., G.A. Padbury, and C.T. Stushnoff. 1998. The ecoregions of Saskatchewan. Canadian Plains Research Centre, Regina, Saskatchewan.

AECB (Atomic Energy Control Board). 1995. Canada: Living with Radiation.

AEP (Alberta Environment and Parks). 2016. Alberta Tier 2 Soil and Groundwater Remediation Guidelines. Land and Forestry Policy Branch, Policy Division.

AEP (Alberta Environment and Parks). 2019. Alberta Tier 1 Soil and Groundwater Remediation Guidelines. Land Policy Branch, Policy and Planning Division.

AIHC (American Industrial Health Council). 1992. Improving risk characterization. In Proceedings of a Symposium, 26-27 September, 1991. Washington, D.C.

Amiro, B.D. 1997. Radiological dose conversion factors for generic non-human biota used for screening potential ecological impacts. Journal of Environmental Radioactivity 35(1):37–51.

Anderson, R. 2002. Castor canadensis, Animal Diversity Web. http://animaldiversity.org/accounts/Castor_canadensis/ (accessed March 21, 2016).

Anderson, S.L., and F.L. Harrison. 1966. Effects of Radiation on Aquatic Organisms and Radiobiological Methodologies for Effects Assessment. Washington, DC: U.S. EPA Report no. 5201/85-016.

ATSDR (Agency for Toxic Substances and Disease Registry). 2004. Toxicological profile for cobalt. U.S. Department of Health and Human Services, Public Health Service, April.

ATSDR (Agency for Toxic Substances and Disease Registry). 2013. Toxicological profile for uranium. U.S. Department of Health and Human Services, Public Health Service, February.

Banfield, A.W.. 1974. The Mammals of Canada. University of Toronto Press, Toronto, Canada.

Beresford, N.A., J. Brown, D. Copplestone, J. Garnier-Laplace, B. Howard, C.M. Larsson, D. Oughton, G. Pröhl, and I. Zinger. 2007. An Integrated Approach to the assessment and management of environmental risks from ionising radiation. Description of purpose, methodology and application. EC project contract no. FI6R-CT-2004-508847. www.erica-project.org.

Bird, G.A., P.A. Thompson, C.R. MacDonald, and S.C. Sheppard. 2002. Ecological Risk Assessment Approach for the Regulatory Assessment of the Effects of Radionuclides Released from Nuclear

Facilities. Presented at the Third International Symposium on the Protection of the Environment from Ionising Radiation. Darwin, Northern Territory, Australia, 22 to 26 July.

Blaylock, B.G., M.L. Frank, and B.R. O'Neal. 1993. Methodology for Estimating Radiation Dose Rates to Fresh Water Biota Exposed to Radionuclides in the Environment. Oak Ridge, TN: ES/ER/TM-78 ORNL.

Blaylock, B.G., and J.R. Trabalka. 1978. Evaluating the Effects of Ionizing Radiation on Aquatic Organisms. Edited by J. T. Lett and H. Adler. Advances in Radiation Biology 7. Advances in Radiation Biology:103–152.

Burnett-Seidel, C., and K. Liber. 2012. Evaluation of sediment quality guidelines derived using the screening-level concentration approach for application at uranium operations in Saskatchewan, Canada. Environmental Monitoring and Assessment 184(3):1593–1602.

Burnett-Seidel, C., and K. Liber. 2013. Derivation of no-effect and reference-level sediment quality values for application at Saskatchewan uranium operations. Environmental Monitoring and Assessment 185(11):9481–9494.

Bush, E., and D.S. Lemmen. 2019. Canada's Changing Climate. Edited by Government of Canada.

Canadian Wildlife Federation. 1993. Hinterlands Who's Who: Red Fox. http://www.hww.ca/en/wildlife/mammals/red-fox.html (accessed December 3, 2019).

CARB (California Air Resources Board). 2008. Staff Report: Methodology for Estimating Premature Deaths Associated with Long-term Exposures to Fine Airborne Particulate Matter in California. California Air Resources Board.

CCME (Canadian Council Of Ministers of the Environment). 2006. A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines.

CCME (Canadian Council Of Ministers of the Environment). 2007. A protocol for the derivation of water quality guidelines for the protection of aquatic life. http://www.ccme.ca/assets/pdf/protocol_aql_2007e.pdf.

CCME (Canadian Council Of Ministers of the Environment). 2012. Canadian Ambient Air Quality Standards (CAAQS). Available on-line: https://ccme.ca/en/air-quality-report.

CCME (Canadian Council Of Ministers of the Environment). 2017. Canadian environmental quality guidelines summary table. http://st-ts.ccme.ca/en/index.html.

CCME (Canadian Council Of Ministers of the Environment). 2019. Canadian Environmental Quality Guidelines. https://www.ccme.ca/en/resources/canadian_environmental_quality_guidelines/.

CEPA/FPAC WGAQOG (Canadian Environmental Protection Act Federal Provincial Advisory Committee Working Group on Air Quality Objectives and Guidelines). 1998. National Ambient Air Quality Objectives for Particulate Matter Executive Summary: Part 1 - Science Assessment Document. Cat. No. H46-2/98-220.

Chambers, D.B., R. V Osborne, and A.L. Garva. 2006. Choosing an alpha radiation weighting factor for doses to non-human biota. Journal of Environmental Radioactivity 87(1):1–14.

Chipman, W.A. 1972. Ionizing Radiation. In Marine Ecology, 1, edited by O. Kinne. Chichester, U.K.: John Wiley and Sons, Ltd.

CNSC (Canadian Nuclear Safety Commission). 2014. Cigar Lake Operation licence conditions handbook, Revision 1. UML-MINE-CIGAR.00/2021, effective January 23, 2014.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2008. COSEWIC assessment and update status report on the short-eared owl *Asio flammeus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2010. COSEWIC Assessment and Status Report on the Yellow Rail *Coturnicops noveboracensis* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. https://wildlife-species.canada.ca/species-risk-registry/virtual_sara/files/cosewic/sr_Yellow Rail_0810_e.pdf.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2013. COSEWIC assessment and status report on the little brown bat *Myotis lucifugus*, northern myotis *Myotis septentrionalis*, tri-colored bat *Perimyotis subflavus* in Canada. Committee on the Status of Endangered Wildlife iln Canada. Ottawa. https://www.registrelep-sararegistry.gc.ca/virtual_sara/files/cosewic/sr_Little Brown Myotis%26Northern Myotis%26Tri-colored Bat_2013_e.pdf.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2014a. COSEWIC assessment and status report on the Gypsy Cuckoo Bumble Bee *Bombus bohemicus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. ix + 56 pp.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2014b. COSEWIC assessment and status report on the wolverine *Gulo gulo* in Canada. https://www.sararegistry.gc.ca/virtual_sara/files/cosewic/sr_Wolverine_2014_e.pdf.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2015. COSEWIC assessment and status report on the Yellow-banded Bumble Bee *Bombus terricola* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. ix + 60 pp. www.registrelep-sararegistry.gc.ca/default_e.cfm.

CSA (Canadian Standards Association). 2012. N288.6-12: Environmental risk assessments at Class I nuclear facilities and uranium mines and mills. June. Reaffirmed 2017.

Davé, N., T. Lim, D. Horne, Y. Boucher, and R. Stuparyck. 1997. Water Cover on Reactive Tailings and Wasterock: Laboratory Stuides of Oxidation and Metal Release Characteristics. In VI International Conference on Acid Rock Drainage, Vancouver, BC, Vol. 2:779-795.

Davis, R.D., P.H.T. Beckett, and E. Wollan. 1978. Critical levels of twenty potentially toxic elements in young spring barley. Plant and Soil 49(2):395–408.

DeForest, D.K., G. Gilron, S.A. Armstrong, and E.L. Robertson. 2012. Species sensitivity distribution evaluation for selenium in fish eggs: Considerations for development of a Canadian tissue-based guideline. Integrated Environmental Assessment and Management 8(1):6–12.

EC/HC (Environment Canada/Health Canada). 2003. Priority Substances List Assessment Report: Releases of Radionuclides from Nuclear Facilities (Impact on Non-Human Biota).

EC/HC (Environment Canada/Health Canada). 2011. Screening assessment for the Challenge: Cobalt, cobalt chloride, sulfuric acid (cobalt (2+) salt (1:1)), sulfuric acid (cobalt salt). January.

EC (Environment Canada). 1997. Environmental assessments of priority substances under the Canadian Environmental Protection Act: Guidance Manual Version 1.0. March.

EC (Environment Canada). 2012. Federal contaminated sites action plan (FCSAP) ecological risk assessment guidance. Prepared for Environment Canada by Azimuth Consulting Group, March.

EC (Environment Canada). 2015. Management plan for the rusty blackbird (*Euphagus carolinus*) in Canada.

EC (Environment Canada). 2017. Federal Environmental Quality Guidelines- Cobalt.

EC (Environment Canada). 2019. Federal environmental quality guidelines. https://www.canada.ca/en/health-canada/services/chemical-substances/fact-sheets/federal-environmental-quality-guidelines.html (accessed December 18, 2019).

ECCC (Environment and Climate Change Canada). 2021a. Federal Environmental Quality Guidelines. Copper. Canadian Environmental Protection Act, 1999. May.

ECCC (Environment and Climate Change Canada). 2021b. Federal environmental quality guidelines: selenium. Environment and Climate Change Canada. Canadian Environmental Protection Act, 1999. June.

Egami, N. 1980. Radiation Effects on Aquatic Organisms. In Proceedings of the International Symposium on Radiation Effects on Aquatic Organisms, Japan, 1979, edited by N. Egami. Scientific Societies Press, Tokyo and University Park Press, Baltimore.

ENSR (ENSR Corporation). 2004. Development of a Standardized Approach for Assessing Potential Risks to Amphibians Exposed to Sediment and Hydric Soils. Prepared for the Naval Facilities Engineering Command (NAVFAC).

FAO and WHO (Food and Agriculture Organization and World Health Organization). 2011. Safety evaluation of certain food additives and contaminants. WHO Food Additives Series: 64. Prepared by the seventy-third meeting of the Joint FAO/WHO Expert Committee on Food Additives (JECFA).

Feugier, A., S. Frelon, P. Gourmelon, and M. Claraz. 2008. Alteration of mouse oocyte quality after a subchronic exposure to depleted uranium. Reproductive Toxicology 26:273–277.

Giesy, J.P., K.R. Solomon, J.R. Coats, K.R. Dixon, J.M. Giddings, and E.E. Kenaga. 1999. Chlorpyrifos: ecological risk assessment in North American aquatic environments. Reviews of Environmental Contamination and Toxicology 160:1–129.

Godfrey, W.E. 1986. The birds of Canada. Revised ed. Ottawa: National Museums of Canada.

Government of Canada. 2020. Species at Risk Public Registry. http://www.sararegistry.gc.ca/sar/index/default_e.cfm.

Grasty, R.L. 1991. A cross-Canada outdoor radon survey. In Fifth International Symposium on the Natural Radiation Environment (NRE-V). Book of Abstracts, 149. https://inis.iaea.org/search/search.aspx?orig_q=RN:23074723.

GS (Government of Saskatchewan). 2019. Saskatchewan Environmental Quality Guidelines. https://envrbrportal.crmp.saskatchewan.ca/seqg-search/ (accessed December 18, 2019).

GS (Government of Saskatchewan). n.d. Woodland Caribou in Saskatchewan. https://publications.saskatchewan.ca/api/v1/products/79478/formats/89811/download.

Hatler, D.F., and A.M.M. Beal. 2003. Furbearer Management Guidelines: Beaver. http://www.env.gov.bc.ca/fw/wildlife/trapping/docs/beaver.pdf (accessed March 21, 2016).

Havens, A. 2006. "Myotis lucifugus" (On-line), Animal Diversity Web. http://animaldiversity.org/accounts/Myotis_lucifugus/ (accessed November 5, 2020).

Health Canada. 1994. Human health risk assessment for priority substances. Cat. No. En40-215/41E. Ottawa, Ontario.

Health Canada. 2000. Canadian Guidelines for the Management of Naturally Occurring Radioactive Materials (NORM).

Health Canada. 2010. Federal contaminated site risk assessment in Canada, Part II: Health Canada toxicological reference values (TRVs) and chemical-specific factors, version 2.0. September.

Health Canada. 2012. Federal contaminated site risk assessment in Canada, Part I: Guidance on human health preliminary quantitative risk assessment (PQRA). Version 2.0.

Health Canada. 2019. Guidelines for Canadian drinking water quality - Summary table. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Health Canada. 2021. Federal contaminated site risk assessment in Canada: toxicological reference values (TRVs). Version 3.0.

IAEA (International Atomic Energy Agency). 1976. Effects of Ionizing Radiation on Aquatic Organisms and Ecosystems. Technical Report Series No. 172. STI/DOC/010/172.

IAEA (International Atomic Energy Agency). 1992. Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards. Technical Reports Series No. 332.

ICRP (International Commission on Radiological Protection). 1996. Age-dependent Doses to Members of the Public from Intake of Radionuclides: Parts Compilation of Ingestion and Inhalation Dose Coefficients. Oxford: ICRP Publication 72. Vol. 26, No. 1, Elsevier Science Ltd.

ICRP (International Commission on Radiological Protection). 2007. The 2007 Recommendations of the International Commission on Radiological Protection. Annals of the ICRP. Publication 103.

Knowles, J. 2001. An investigation into the effects of chronic radiation on fish. R&D Technical Report P3-053/TR. The Centre for Environment, Fisheries & Aquaculture Science, 1–42.

Langmuir, D., P. Chrostowski, B. Vigneault, and R. Chaney. 2004. Issue paper on the environmental chemistry of metals. Submitted to U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. ERG, Lexington, MA.

Linares, V., M.L. Albina, M. Bellés, E. Mayayo, D.J. Sánchez, and J.L. Domingo. 2005. Combined action of uranium and stress in the rat II. Effects on male reproduction. Toxicology Letters 158:186–195.

Llobet, J.M., J.J. Sirvent, A. Ortega, and J.L. Domingo. 1991. Influence of chronic exposure to uranium on male reproduction in mice. Fundamental and Applied Toxicology 16:821–829.

Lowe, L.M. 1997. Impact of the 235U series on doses from intakes of natural uranium and decay progeny. Health Physics 73(4):694–695.

Makeyeva, A.P., N.G. Yemel'yanova, N.V. Beolva, and I.N. Ryabov. 1995. Radiobiological analysis of silver carp, *Hypophthalmichthys molitrix*, from the cooling pond of Chrernobyl nuclear power plant since the time of the accident. Journal of Ichthyology 35(2):40–62.

Maynard, E.A., W.L. Down, and H.C. Hodge. 1953. Oral toxicity of uranium compounds. In Pharmacology and Toxicology of Uranium Compounds, edited by C. Voegtlin and H. C. Hodge, 1221–1369. New York, NY: McGraw-Hill.

Maynard, E.A., and H.C. Hodge. 1949. Studies of the toxicity of various uranium compounds when fed to experimental animals. In Pharmacology and Toxicology of Uranium Compounds, National Nuclear Energy Series (VI), edited by I. C. Voegtlin and H. C. Hodge, 309–376. New York, NY: McGraw-Hill.

McBride, M.B. 1994. Environmental Chemistry of Soils. New York, NY: Oxford University Press Inc.

Minister of Supply and Services Canada. 1996. A national ecological framework for Canada. Written and compiled by the Ecological Stratification Working Group (Centre for Land and Biological Resources Research, Research Branch, Agriculture and Agri-Food Ca.

Morland, S. 2011. Lagopus lagopus, Animal Diversity Web. http://animaldiversity.org/accounts/Lagopus_lagopus/ (accessed March 22, 2016).

Mothersill, C. 2015. Personal communication, Dr Carmel Mothersill , McMaster University, February.

Nagorsen, D.W. 2005. Rodents and lagomorphs of British Columbia. Royal BC Museum Handbook: Volume 4 of the Mammals of British Columbia.

NCRP (National Council on Radiation Protection). 1991. Effects of Ionizing Radiation on Aquatic organisms. Report No. 109.

NRCC (National Research Council of Canada). 1983. Radioactivity in the Canadian Aquatic Environment. Ottawa, ON: Public. No. NRCC-19250, National Radiation Council of Canada.

NWF (National Wildlife Federation). 2021. Wolverine. National Wildlife Federation. https://www.nwf.org/Educational-Resources/Wildlife-Guide/Mammals/Wolverine (accessed February 17, 2021).

Polikarpov, G.G. 1966. Radioecology of Aquatic Organisms. New York, NY: Reinhold Book Division.

Pröhl, G., J. Brown, J.M. Gomez-Ros, S. Jones, D. Woodhead, J. Vives, V. Taranenko, and H. Thørring. 2003. Dosimetric Models and Data for Assessing Radiation Exposure to Biota. Deliverable 3 to the project "FASSET" Framework for the Assessment of Environmental Impact.

Raimondo, S., P. Mineau, and M.G. Barron. 2007. Estimation of chemical toxicity to wildlife species using interspecies correlation models. Environmental Science & Technology 41(16):5888–5894.

Receveur, O., A. Ing, L. Chan, and H. Kuhnlein. 1998. Recovery of Yellowknives Dene dietary survey. Ste Anne de Bellevue, Quebec, Canada: Centre for Indigenous Peoples' Nutrition and Environment (CINE).

Richardson, G.M., and Stantec Consulting Ltd. 2013. 2013 Canadian Exposure Factors Handbook. Toxicology Centre, University of Saskatchewan, Saskatoon, SK CANADA. Available at: http://www.usask.ca/toxicology/docs/cef.

Richardson, G.M. 1997. Compendium of Canadian human exposure factors for risk assessment. O'Connor Associates Environmental Inc., Ottawa ON.

Rodewald, P. 2020. Birds of the World. https://birdsna.org/Species-Account/bna/home.

Sample, B.E., D.M. Opresko, and G.W. Sutter II. 1996. Toxicological benchmarks for wildlife: 1996 revision. Prepared by the Risk Assessment Program Health Sciences Research Division for the U.S. Department of Energy, Office of Environmental Management.

Saskatchewan Ministry of Labour Relations and Workplace Safety. 2010. Chemical and Biological Substances Guide. Occupational Health and Safety. January.

Schoof, R.A., L.J. Yost, J. Eickhoff, E.A. Crecelius, D.W. Cragin, D.M. Meacher, and D.B. Menzel. 1999. A market basket survey of inorganic arsenic in food. Food and Chemical Toxicology : An International Journal Published for the British Industrial Biological Research Association 37(8):839–846.

SKCDC (Saskatchewan Conservation Data Centre). 2020. Taxa list for vertebrates. http://www.biodiversity.sk.ca/SppList/verts.pdf (accessed June 6, 2018).

SKMOE (Saskatchewan Ministry of Environment). 2012. Saskatchewan Air Quality Modelling Guideline Version : March 2012. March 2012. Technical Resources Branch. Air Science and Monitoring Unit.

Smith, J. 2002. Canis lupus, Animal Diversity Web. http://animaldiversity.org/accounts/Canis_lupus/ (accessed March 21, 2016).

Solomon, K.R., and J.M. Giddings. 2000. Understanding probabilistic risk assessment. SETAC Globe 1:34–35.

Sparling, D.W., G. Linder, C.A. Bishop, and S.K. Krest. 2010. Ecotoxicology of Amphibians and Reptiles. 2nd ed.

van Straalen, N.M., and K. van Leeuwen. 2002. European history of species sensitivity distributions. In Species Sensitivity Distributions in Ecotoxicology, edited by L. Posthuma, G. W. I. I. Suter, and T. P. Traas, 19–34. Lewis Publishers.

Suter, G.W.I. 1993. Ecological risk assessment. Lewis Publishers. Boca Raton.

Templeton, W.L., R.E. Nakatani, and E.E. Held. 1971. Radiation Effects. In Radioactivity in the Marine Environment, 223–239. Washington DC: National Academy of Science.

Thompson, P.A., J. Kurias, and S. Mihok. 2005. Derivation and use of Sediment Quality Guidelines for Ecological Risk Assessment of metals and radionuclides released to the environment from uranium mining and milling activities in Canada. Environmental Monitoring and Assessment 110:71–85.

Trivedi, A., and N. Gentner. 2000. Ecodosimetry Weighting Factor (eR) for Non-Human Biota. In Proceedings for IRP-10, 14-19 May 2000, Hiroshima, Japan. Harmonization of Radiation, Human Life and the Ecosystem.

Tseng, W.P. 1977. Effects and dose-response relationships of skin cancer and blackfoot disease with arsenic. Environmental Health Perspectives.

Tseng, W.P., H.M. Chu, S.W. How, J.M. Fong, C.S. Lin, and S. Yeh. 1968. Prevalence of skin cancer in an endemic area of chronic arsenicism in Taiwan. Journal of the National Cancer Institute 40(3):453–463.

U.K. Environment Agency (United Kingdom Environment Agency). 2002. Impact Assessment of Ionising Radiation on Wildlife. R and D Publication 128.

U.S. DOE (United States Department of Energy). 2002. A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota. DOE Standard DOE-STD-1153-2002.

U.S. EPA (United States Environmental Protection Agency). 1989. Risk Assessment Guidance for Superfund (RAGS) Volume 1 Human Health Evaluation Manual (Part A). December. EPA/540/1-89/002.

U.S. EPA (United States Environmental Protection Agency). 1992. Framework for Ecological Risk Assessment. EPA/630/R-92/001 February.

U.S. EPA (United States Environmental Protection Agency). 1993. Wildlife exposure factors handbook Volume I of II. Office of Health and Environmental Assessment, Office of Research and Development. U.S. Environmental Protection AGency, Washington, DC. EPA/600/R-93/187. December.

U.S. EPA (United States Environmental Protection Agency). 2005. Guidance for developing ecological soil screening levels. February Revision.

U.S. EPA (United States Environmental Protection Agency). 2009. Guidance on the Development, Evaluation, and Application of Environmental Models. EPA/100/K-09/003 | March.

U.S. EPA (United States Environmental Protection Agency). 2016. Aquatic Life Ambient Water Quality Criterion for Selenium - Freshwater. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology. EPA 822-R-16-006, June. https://www.epa.gov/wqc/aquatic-life-ambient-water-quality-criteria.

United States Forest Service. 2011. National Lichens & Air Quality Database and Clearinghouse. An Agency of the United States Department of Agriculture. http://gis.nacse.org/lichenair/index.php.

University of Saskatchewan. 2009. Soils of Saskatchewan. College of Agriculture and Bioresources. http://www.soilsofsask.ca/index.php. Accessed November 10, 2009.

UNSCEAR (United Nations Scientific Committee on the Effects of Atomic Radiation). 1996. Report to the General Assembly, with Scientific Annex.

UNSCEAR (United Nations Scientific Committee on the Effects of Atomic Radiation). 2008a. Sources and Effects of Ionizing Radiation. Annex E. Effects of Ionizing Radiation on Non-Human Biota.

UNSCEAR (United Nations Scientific Committee on the Effects of Atomic Radiation). 2008b. Sources and Effects of Ionizing Radiation. Annex B. Exposures of the public and workers from various sources of radiation.

WHO (World Health Organization). 2000. Air quality guidelines for Europe, second edition. WHO Regional Publications, European Series, No. 91.

WHO (World Health Organization). 2006. WHO Air Quality Guidelines for Particulate Matter, Ozone, Nitrogen Dioxide and Sulfur Dioxide, Global Update 2005. WHO/SDE/PHE/OEH/06.02. https://apps.who.int/iris/bitstream/handle/10665/69477/WHO_SDE_PHE_OEH_06.02_eng.pdf?sequence=1.

WHO (World Health Organization). 2021. WHO global air quality guidelines. Particulate matter (PM2.5 and PM10), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. Geneva: World Health Organization; 2021. Licence: CC BY-NC-SA 3.0 IGO.

Wilson, R., H. Jones-Otazo, S. Petrovic, I. Mitchell, Y. Bonvalot, D. Williams, and G.M. Richardson. 2013. Revisiting dust and soil ingestion rates based on hand-to-mouth transfer. Human and Ecological Risk Assessment: An International Journal 19(1):158–188.

Wilson, R., and M. Richardson. 2012. Proposed toxicological reference values and risk-based soil concentrations for protection of human health from lead (Pb) at Federal contaminated sites. 2012 RPIC Federal Contaminated Sites National Workshop, APril 30 - M ay 3, 2012. Toronto, Ontario.

Woodhead, D.S. 1984. Contamination Due to Radioactive Materials. In Marine Ecology, Vol. V, Part 3 Pollution and Protection of the Seas-- Radioactive Materials, Heavy Metals and Oil, edited by O. Kinne, 1111–1287. New York, NY: John Wiley and Sons.

Yost, L.J., R.A. Schoof, and R. Aucoin. 1998. Intake of Inorganic Arsenic in the North American Diet. Human and Ecological Risk Assessment: An International Journal 4(1):137–152.