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Canadian Environmental Protection Act, 1999

PRIORITY SUBSTANCES LIST ASSESSMENT REPORT

RELEASES OF RADIONUCLIDES FROM NUCLEAR FACILITIES (IMPACT ON NON-HUMAN BIOTA)

Environment Canada
Health Canada

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LIST OF ACRONYMS AND ABBREVIATIONS

a	annum (year)
Ac	actinium
AECB	Atomic Energy Control Board
AECL	Atomic Energy of Canada Limited
As	arsenic
Bi	bismuth
Bq	becquerel
bw	body weight
C	carbon
CANDU	Canadian Deuterium Uranium
Cd	cadmium
Ce	cerium
CEPA	*Canadian Environmental Protection Act*
CEPA 1999	*Canadian Environmental Protection Act, 1999*
CL	confidence limit
CMF	Central Maintenance Facility
CNSC	Canadian Nuclear Safety Commission
Co	cobalt
Cr	chromium
CR	concentration ratio
CRL	Chalk River Laboratories
Cs	cesium
CTV	Critical Toxicity Value
Cu	copper
CWS	Canadian Wildlife Service
d	day
DCF	dose conversion factor
DNA	deoxyribonucleic acid
DSB	double strand break
dw	dry weight
EEV	Estimated Exposure Value
ENEV	Estimated No-Effects Value
eV	electron volt
F	fluorine
f_1	absorption factor
Fe	iron
FIG	Field Irradiation Gamma
g	gram
GM	geometric mean
Gy	gray

h	hour
H	hydrogen
^{3}H	tritium
ha	hectare
HT	tritiated hydrogen gas
HTO	tritiated water (tritium oxide)
I	iodine
I_{water}	intake from water
I_{food}	intake from food
$I_{soil/sediment}$	intake from soil/sediment
IAEA	International Atomic Energy Agency
ICRP	International Commission for Radiological Protection
J	joule
K	potassium
K_d	partition coefficient
kg	kilogram
km	kilometre
L	litre
LC_{50}	median lethal concentration
LD_{50}	median lethal dose
LEL	Lowest Effect Level
LET	linear energy transfer
LOAEL	Lowest-Observed-Adverse-Effect Level
m	milli- (prefix)
m^3	cubic metre
mg	milligram
µ	micro- (prefix)
µm	micrometre
Mn	manganese
MWe	megawatt electrical
Nb	niobium
NCRP	National Council on Radiation Protection and Measurement
NGS	nuclear generating station
Ni	nickel
NOEC	No-Observed-Effect Concentration
NOEL	No-Observed-Effect Level
NRCC	National Research Council of Canada
NRU	National Research Universal
OPG	Ontario Power Generation
Pb	lead
PGWMF	Port Granby Waste Management Facility
Pr	praseodymium

PSL	Priority Substances List
PSL2	Priority Substances List #2
Ra	radium
RBE	relative biological effectiveness
Rn	radon
RQ	risk quotient
Ru	rubidium
RWOS	Radioactive Waste Management Operations Site
Sb	antimony
SEL	Severe Effect Level
SLC	screening-level concentration
Sr	strontium
SSB	single strand break
Sv	Sievert
TC	transfer coefficient
TDI	total daily intake
Th	thorium
TMA	tailings management area
U	uranium
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
WCEM	Wildlife Contaminant Exposure Model
WMF	waste management facility
ww	wet weight
WWMF	Welcome Waste Management Facility
Zn	zinc
Zr	zirconium

SYNOPSIS

The impact of the release of radionuclides from nuclear facilities on non-human biota was assessed. Nuclear facilities include all aspects of the uranium fuel chain, from mining and milling through to power generation and waste management. Although nuclear facilities release non-radioactive substances (e.g., metals, organic chemicals), effects of such non-radioactive substances were not considered in this assessment.

Because of the variety of industrial activities and processes that result in the release of a large number of radionuclides with different radiological half-lives, chemical, biological and environmental properties, sectorial assessments were conducted. The sectors and numbers of facilities considered are: 5 uranium mines and mills, 2 uranium refineries and conversion plants, 3 waste management areas, and 5 nuclear power plants. Heavy water production facilities were not included because there is no production, use or release of radionuclides from these facilities.

Uranium (U) and thorium (Th) and their decay chain daughter radionuclides are the radionuclides of concern released from U mines, whereas U is the main radionuclide of concern released from U refining and conversion facilities. Radionuclides of concern in tailings management areas are primarily radium-226 (^{226}Ra) and U, although other radionuclides (e.g., ^{3}H, ^{14}C, ^{60}Co, ^{90}Sr and ^{137}Cs) may also be important in some waste management areas.

Fission and activation products released from nuclear generating stations include ^{3}H, ^{14}C, ^{51}Cr, ^{54}Mn, ^{59}Fe, ^{60}Co, ^{65}Zn, ^{90}Sr, ^{95}Zr, ^{106}Ru, ^{124}Sb, $^{124\text{-}135}$I, ^{137}Cs and ^{144}Ce. Releases of radionuclides from these facilities are primarily to air or to water. Emissions to air will result in deposition of particle reactive radionuclides and increased scavenging of radionuclides from the plume with distance from the source. Mobile radionuclides such as the inert gases will disperse quickly and reach background concentrations a short distance (few km) from the source. Most of the radionuclides released are particle reactive and partition either from water to sediment or from air to soil.

There are two modes of toxic action for the releases being assessed; one is the chemical toxicity of the elements released, and the other is their radiological toxicity of the radionuclides released. Only U has the potential to result in chemical toxicity. Chemical toxicity is the normal mode of toxic effect for environmental contaminants. Radiotoxicity differs in that radiation dose, the measure of radiation, results from radionuclides incorporated in tissues (internal dose) and from external radionuclides (external dose) that emit radiation adjacent to the organism.

For the chemical toxicity of U, releases are largely restricted to the front and back ends of the nuclear fuel chain, the mining and milling of the U and the waste management. Comparison of realistic exposure values to estimated no effects values indicates potential for localized harm for a number of organisms at several of the mining and milling areas and two of the three waste management areas examined.

Risks to biota from exposure to ionizing radiation are also largest associated with the mining and milling of U, where comparison of realistic exposure and no effects values suggests that localized harm may be occurring at some facilities. Ionizing radiation released from the power reactors is not expected to cause environmental harm. Releases from two waste management sites also are not expected to result in exposure of biota to harmful amounts ionizing radiation. Biota may however be harmed by exposure to ionizing radiation at one of the waste management areas examined. Current releases of ionizing radiation from uranium refineries and conversion plants are not expected to cause environmental harm. Although, it is possible that ionizing radiation is harming aquatic organisms near one U refinery, this is likely attributable to waste management and operational practices of past decades.

Contributions to ground level ozone formation, stratospheric ozone depletion and climate change by radionuclides released to the atmosphere from nuclear facilities are negligible.

Based on available data concerning the effects from exposure to both uranium and ionizing radiation, it has been concluded that (i) releases of radionuclides from uranium mines and mills and waste management areas are entering the environment in quantities or concentrations or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity; (ii) releases of radionuclides from uranium refineries and conversion facilities and power and research reactors are not entering the environment in quantities or concentrations or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity; and (iii) releases of radionuclides from nuclear facilities are not entering the environment in quantities or concentrations or under conditions that constitute or may constitute a danger to the environment on which life depends. Therefore, it is proposed that releases of radionuclides from uranium mines and mills and waste management areas be considered "toxic" as defined in Section 64 of the *Canadian Environmental Protection Act, 1999* (CEPA 1999).

It is recommended that investigations of options to reduce exposure to releases of radionuclides from U mines and mills and waste management areas be considered a high priority. Discussions have been initiated with the Canadian Nuclear Safety Commission (CNSC) (formerly the Atomic Energy Control Board [AECB]) to determine whether it will be possible to manage these releases under the new *Nuclear Safety and Control Act*. It is proposed that the process for risk management could be formalized in the memorandum of understanding currently being negotiated between Environment Canada and the CNSC.

Further research into the estimated no effects values for exposure of non-human biota to radiation should be a priority. This should include research into the genetic effects of environmentally relevant radiation doses and into the effects of alpha-emitters on ecologically relevant endpoints, for the purpose of verifying the appropriateness of the weighting factor to account for the greater relative biological effectiveness of alpha-emitters. In PSL2 assessments, data on genetic damage were not taken into

consideration in the derivation of estimated-no-effects values because of the difficulty in interpreting the significance of these effects at the population level (i.e. population fitness and survival). Therefore, priority should also be given to research on the ecological significance of genetic effects and their consideration in the ecological risk assessment of radioactive and non-radioactive environmental contaminants.

1.0 INTRODUCTION

The *Canadian Environmental Protection Act, 1999* (CEPA 1999) requires the federal Ministers of the Environment and of Health to prepare and publish a Priority Substances List (PSL) that identifies substances, including chemicals, groups of chemicals, effluents and wastes, that may be harmful to the environment or constitute a danger to human health. The Act also requires both Ministers to assess these substances and determine whether they are "toxic" or capable of becoming "toxic" as defined in Section 64 of the Act, which states:

> ...a substance is toxic if it is entering or may enter the environment in a quantity or concentration or under
> conditions that
> (a) have or may have an immediate or long-term harmful effect on the environment or its biological diversity;
> (b) constitute or may constitute a danger to the environment on which life depends; or
> (c) constitute or may constitute a danger in Canada to human life or health.

Substances that are assessed as "toxic" as defined in Section 64 may be placed on Schedule I of the Act and considered for possible risk management measures, such as regulations, guidelines, pollution prevention plans or codes of practice to control any aspect of their life cycle, from the research and development stage through manufacture, use, storage, transport and ultimate disposal.

Based on initial screening of readily accessible information, the rationale for assessing releases of radionuclides from nuclear facilities (impact on non-human biota) provided by the Ministers' Expert Advisory Panel on the Second Priority Substances List (Ministers' Expert Advisory Panel, 1995) was as follows:

> The Panel notes that while the Atomic Energy Control Board (AECB) already assesses the risk to human health of radionuclides released from nuclear facilities, there are gaps in the assessment of the risks to non-human species. The Panel appreciates that such an assessment will undoubtedly be complex and will require the expert assistance of AECB; nonetheless, it is convinced that the potential risks to non-human species warrant an ecological assessment under CEPA.

A description of the approaches to assessment of the effects of Priority Substances on the environment is available in a published companion document. The document, entitled "Environmental Assessments of Priority Substances under the *Canadian Environmental Protection Act*. Guidance Manual Version 1.0 — March 1997" (Environment Canada, 1997), provides guidance for conducting environmental assessments of Priority Substances in Canada. This document may be purchased from:

Environmental Protection Publications
Environmental Technology Advancement Directorate
Environment Canada
Ottawa, Ontario
K1A 0H3

It is also available on the Commercial Chemicals Evaluation Branch web site at www.ec.gc.ca/cceb1/eng/psap.htm under the heading "Technical Guidance Manual." It should be noted that the approach outlined therein has evolved to incorporate recent developments in risk assessment methodology, which will be addressed in future releases of the guidance manual for environmental assessments of Priority Substances.

The search strategy employed in the identification of data relevant to assessment of potential effects on the environment involved determination of environmental concentrations, based on licensee annual reports and environmental impact statements, as well as harmful concentrations of chemicals or radiation doses. Review articles were consulted where appropriate. However, all original studies that form the basis for determining whether releases of radionuclides from nuclear facilities are "toxic" under CEPA 1999 have been critically evaluated by staff of the Canadian Nuclear Safety Commission (CNSC) (formerly the Atomic Energy Control Board [AECB]) for Environment Canada.

This Assessment Report was produced by G. Bird and P. Thompson, CNSC. A first draft of the sections of the supporting documentation dealing with uranium (U) toxicity and effects of radiation on wildlife was prepared by C. Macdonald, Northern Environmental Consulting and Analysis (Macdonald, 1998, 1999). A first draft of sections on the effects of radiation on plants and the fate and behaviour of radionuclides in the environment was prepared by M. Sheppard and S. Sheppard, ECOMatters Inc. (ECOMatters Inc., 1999a,b).

An Environmental Resource Group was established by Environment Canada to assist in the review of the environmental assessment of releases of radionuclides. The Environmental Resource Group, which consisted of scientific experts from industry and consulting firms, was established in the fall of 1996. Members included:

J. Cornett, Atomic Energy of Canada Limited (AECL)
N. Garisto, Senes Consultants Limited
F. Harrison, Lawrence Livermore Laboratories
C. Macdonald, Northern Environmental Consulting and Analysis
S. Sheppard, ECOMatters Inc.

This Assessment Report and the supporting documentation (Bird *et al*., 2000) were reviewed by members of the Environmental Resource Group as well as by Pat Doyle, Environment Canada.

The entire Assessment Report was reviewed and approved by the Environment Canada/Health Canada CEPA Management Committee.

The text of the Assessment Report has been structured to address environmental effects relevant to determination of "toxic" under Paragraphs 64(a) and (b).

Copies of this Assessment Report are available upon request from:

Inquiry Centre
Environment Canada
Main Floor, Place Vincent Massey
351 St. Joseph Blvd.
Hull, Quebec
K1A 0H3

or on the Internet at:

www.ec.gc.ca/cceb1/eng/public/index_e.html

Unpublished supporting documentation, which presents additional information, is available upon request from:

Commercial Chemicals Evaluation Branch
Environment Canada
14th Floor, Place Vincent Massey
351 St. Joseph Blvd.
Hull, Quebec
K1A 0H3

or

Canadian Nuclear Safety Commission
P.O. Box 1046, Station B
280 Slater Street
Ottawa, Ontario
K1P 5S9

2.0 IDENTITY AND ENTRY CHARACTERIZATION

Nuclear facility is a broad term that encompasses a variety of industrial activities and processes that result in the release of a large number of radionuclides with different radiological half-lives and chemical, biological and environmental properties. Sectorial assessments were carried out to reduce the complexity of the ecological assessment and to provide more valuable information to risk managers. The sectors for which radionuclide releases were assessed were (a) U mines and mills, (b) U refining and conversion facilities and fuel fabrication facilities (i.e., U decay series radionuclides), (c) power and research reactors (fission and activation products) and (d) waste management facilities. Heavy water production plants, mentioned in the Ministers' Expert Advisory Panel (1995) report, are not included in this assessment, since they do not use, produce or release radionuclides.

Processes included in this assessment are mining of U ore, milling of the ore to produce yellowcake, U refining and conversion, CANDU (Canadian Deuterium Uranium) fuel fabrication, nuclear power generation and nuclear waste management. The following sections describe the production, use and environmental releases (entry characterization) of U and U decay chain radionuclides during mining, refinery and fabrication processes as well as releases of activation and fission products from nuclear generating stations (NGSs) to the environment. A review of the behaviour and fate of U, U decay chain radionuclides, activation products and fission products in the environment is given in the supporting document (Bird *et al*., 2000).

Substances other than radionuclides are also released from nuclear facilities. For example, U mines and mills release heavy metals (e.g., cadmium [Cd], nickel [Ni], copper [Cu] and arsenic [As]) and saline solutions into the environment. Waste management facilities (WMFs) may release organic contaminants and heavy metals. Nuclear generating stations release hydrazine and metals such as Cu and zinc (Zn). The effects of non-radioactive contaminants released by nuclear facilities to the environment were not considered in the present assessment.

This assessment is directed towards radionuclides released from nuclear facilities. In several cases, especially for mine and mill tailings, natural biota are present inside the facility boundaries — for example, in a tailings management area (TMA). In this assessment, only releases from the facilities are considered. In other words, the assessment would not consider exposure resulting from consumption of, for example, blueberries growing on a TMA, but would include wetland plants growing in a wetland downstream from the tailings.

2.1 Uranium mines and mills

Uranium mills extract triuranium octoxide (U_3O_8) from crushed, ground ores by either an acid or alkaline leaching process. After the leaching process, the resultant solution containing U goes through a solvent extraction process in which the U is purified and concentrated. More than 90% of the U in the ore is recovered in the milling process. Yellowcake, the end product of the milling process, is then shipped to

a U refinery. The residual (tailings), which represents crushed ore minus most of the U, is pumped as a slurry to a tailings management facility. Uranium tailings contain up to 85% of the radioactivity initially present in the U ore, since most of the U decay products remain in the tailings (Landa and Gray, 1995).

Twelve mines and 11 mills operated near Elliot Lake, Ontario, between 1955 and 1959. The decline in the demand for U resulted in the closure of the last operating mine in 1996. The first U mill in northern Saskatchewan started operations in 1953 at Beaverlodge Lake. Today, there are four operating mines and two mining projects in Saskatchewan that have undergone environmental assessment under the *Canadian Environmental Assessment Act* but are not yet in operation. There are also five decommissioned or abandoned mines in northern Saskatchewan.

Uranium and thorium (Th) decay chain radionuclides are released to the environment from U mining and milling operations. Thorium-232 and ^{238}U, the parent isotopes of the Th and U decay chains, are the most abundant isotopes of U and Th in the earth's crust; on average, there are about 4 mg $U \cdot g^{-1}$ and 10 mg $Th \cdot g^{-1}$ in the earth's crust (Whicker and Schultz, 1982). The relative proportion of U and Th decay chain radionuclides released to the environment will be similar to the relative occurrence of Th in the U ores being mined and milled. The decay chains associated with U and Th are presented in Figure 1.

Uranium mill tailings deposited on land are relatively dry, and wind erosion of dust may suspend particulates contaminated with U, ^{226}Ra, ^{210}Pb and other decay products. Dust particles from tailings and from mill vents constitute a potential exposure pathway for terrestrial biota through inhalation. Wet and dry deposition may increase radionuclide concentrations in surrounding soils and vegetation. Tailings are also a point source of radionuclides released into groundwater. Leaching of ^{226}Ra from the tailings may contaminate surrounding groundwater and surface water.

Radionuclides also enter the aquatic environment as a result of releases of treated mill effluent. Generally, the effluent treatment plants also treat contaminated water collected from tailings management facilities. Treated effluent quality is routinely monitored. During the ore milling process, radionuclides are also released to the atmosphere through various stacks. These emissions are routinely monitored. Information on these releases is presented in the supporting document (Bird *et al*., 2000).

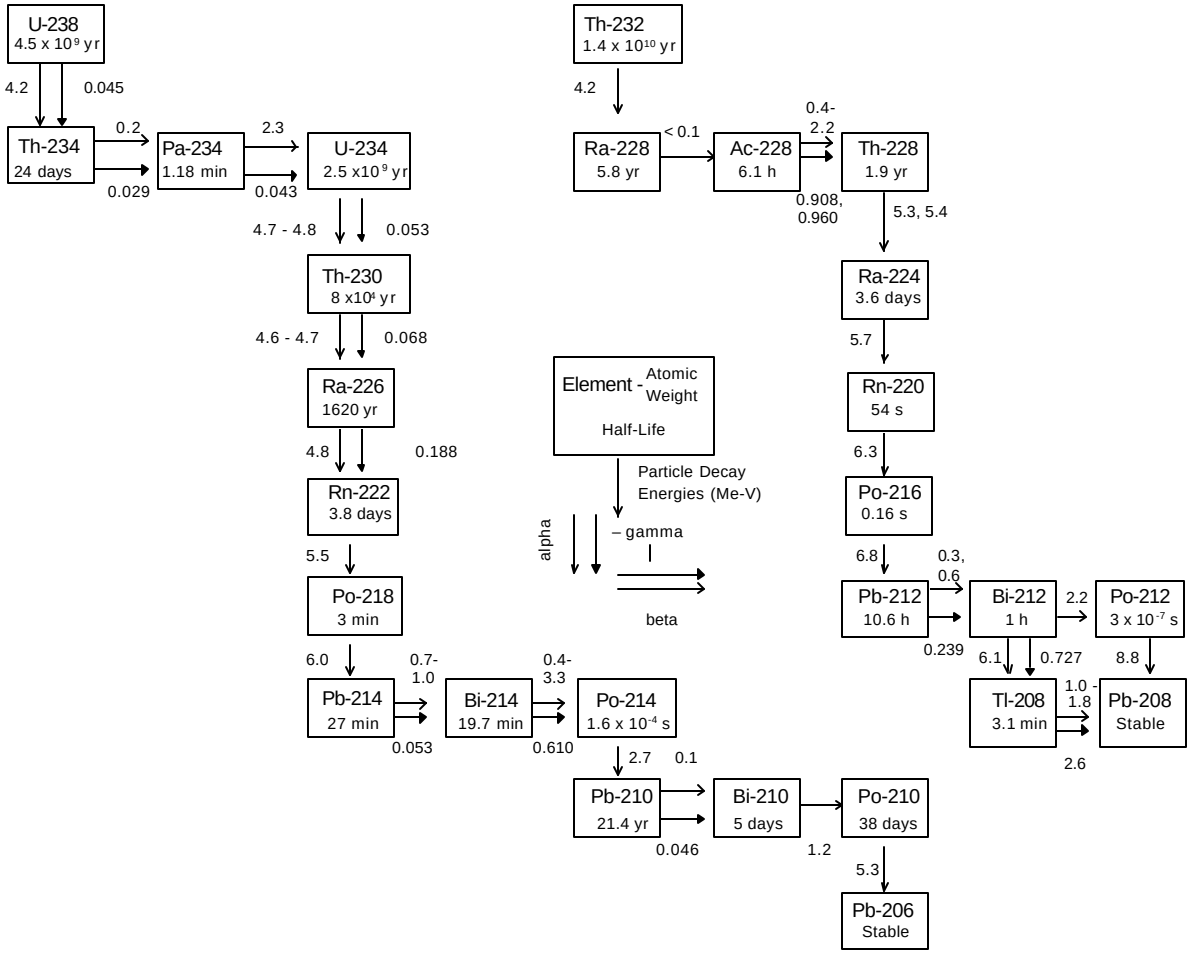

Figure 1: Uranium-238 and ^{232}Th decay chain radionuclides with half-lives and mode of decay (Baweja *et al.*, 1987)

2.2 Uranium refining and conversion facilities

The major processing steps involved in U refining and conversion are the purification of yellowcake from U mills to uranium trioxide (UO_3) in the UO_3 circuit and conversion of UO_3 to uranium hexafluoride (UF_6) in the UF_6 circuit. It is in the form of UF_6 that U is shipped to foreign markets. In another processing stage, the uranium dioxide (UO_2) circuit, an intermediate product from the UO_3 circuit is converted to reactor-grade UO_2 for use in CANDU fuel. Finally, U metal is produced from uranium tetrafluoride (UF_4) drawn from the UF_6 circuit.

There are two U refining and conversion facilities operating in Ontario. Another facility, located in Alberta, closed in 1987 but is not completely decommissioned (AECB, 1996).

Releases that occur at different stages of the U refining and conversion process are collected and vented through a common stack. Atmospheric releases of U occur in different sections of a UO_3 plant (e.g., warehouse, UO_3 circuit). There is also a potential for releases of ^{232}Th, ^{230}Th, ^{226}Ra and ^{222}Rn. Atmospheric releases of U also occur from the UF_6 and UO_2 plants. Both soluble and insoluble compounds are released from the UF_6 plant, whereas all U released from the UO_2 plant is assumed to be insoluble. Intermittent production of U metal results in intermittent atmospheric emissions of both soluble and insoluble U compounds during the production process (Environment Canada, 1984). Data on releases of U to the atmosphere are presented in the supporting document (Bird *et al*., 2000).

The U refining and conversion process also results in U release to the aquatic environment, usually through a single effluent discharge point. Uranium is collected in the liquid effluent from the UO_3, UF_6 and UO_2 plants. The U metal plant has not been identified as a source of U to the aquatic environment (Environment Canada, 1984). Data on releases to the aquatic environment and to municipal sewers are presented in the supporting document (Bird *et al*., 2000).

Cameco carries out the refining and conversion processes in its facilities in Blind River and Port Hope, Ontario. At Blind River, yellowcake is made into UO_3. The UO_3 from Blind River is shipped to the Port Hope conversion facility, where the UO_3 is converted into UO_2 for domestic reactor fuel production and to UF_6 for export. Uranium metal is also produced in a Speciality Metal Plant at Port Hope (AECB, 1998).

Three facilities, all in Ontario, are licensed by the CNSC for the fabrication of CANDU fuel pellets and/or fuel bundles: one in Port Hope, the second in Toronto and the third in Peterborough. Uranium emissions to the atmosphere are continuously monitored at the point of release of all process stacks. Liquid effluent is discharged to the sanitary sewer system.

2.3 Nuclear generating stations

Nuclear power plants are located in Ontario, Quebec and New Brunswick. These are CANDU reactors, which use natural U fuel. Heavy water is used as the moderator and primary coolant (NRCC, 1983). Nuclear power reactors produce radionuclides either as a result of the fission of U atoms (i.e., fission products) in the fuel or as the result of the absorption of neutrons by the coolant and the structural components (e.g., pressure tubes) of the reactor (i.e., activation products). Nuclear power plants routinely report total annual ($Bq \cdot a^{-1}$) atmospheric and liquid releases of radionuclides to the CNSC as part of their licence conditions. They also perform environmental monitoring in the area surrounding their facility and report the data in their annual environmental radiological data reports.

2.4 Nuclear waste management facilities

Waste management facilities for radioactive wastes were developed to handle a variety of wastes, including contaminated soils (historical contamination), liquid and solid wastes, contaminated filters and resins from pollution control systems of nuclear power plants, and reactor spent fuel. Some facilities are well-engineered recent structures, while others are older, less well designed sites. There are several facilities located in Ontario (i.e., Port Granby Waste Management Facility and Welcome Waste Management Facility [WWMF], both near Port Hope, Chalk River Laboratories [CRL] near Chalk River, and Bruce Radioactive Waste Operations Site 1 and 2), one facility in Gentilly, Quebec, and one in Point Lepreau, New Brunswick.

3.0 EFFECTS AND EXPOSURE CHARACTERIZATION AND RISK ANALYSIS

The overall approach, common to all of the PSL assessments, follows the guidelines in Environment Canada (1997). The central concept is to characterize ecological risk as a quotient of the Estimated Exposure Value (EEV) divided by the Estimated No-Effects Value (ENEV), where the value is concentration or dose. If this risk quotient (RQ) is less than unity, there is some assurance that the contaminant is not toxic. Obviously, this conclusion is highly dependent on the values chosen for the EEV and ENEV. To place emphasis on the contaminants most likely to be toxic, or on facilities where the contaminants are most likely to be toxic, this assessment proceeded in two stages. First, conservative assumptions and data are used so that a conservative RQ less than unity indicates very low probability of the harmful effect. This is referred to as a Tier 1 assessment (Environment Canada, 1997). If the conservative RQ is greater than unity, then a more realistic assessment is invoked, called a Tier 2 assessment (Environment Canada, 1997). In the realistic assessment, EEVs are computed as realistic values. The present assessment does not proceed to a probabilistic assessment, as enough information is available to provide realistic estimates of exposure (EEV), and there is no benefit to performing stochastic simulations to estimate concentrations in the environment and biota when these are already known (measured).

The conservative EEV is computed in this assessment as the highest concentration observed or reasonably expected in the relevant environmental media. For example, the conservative EEV for concentrations in water might be set equal to the highest concentration measured during the monitoring of receiving waters near a facility, even if it is from a very localized area of contamination or from a specific event. Realistic EEVs, as the name implies, are more realistic and, for example, would be the mean concentration in the receiving waters, averaged over space and time.

The ENEV is intended to represent endpoints that clearly have ecological relevance and that are reasonable and realistic. The ENEV was set with as much rigour as possible and is based on literature effects data. The same ENEV was used for both conservative and realistic assessments.

A distinct feature of this assessment, among the PSL2 assessments, is that there are two very different modes of toxic effect. The first is radiotoxicity, the impact of ionizing radiation resulting from the radioactive decay of the released radionuclides. Ionizing radiation has distinct biological and biochemical effects and can impact organisms from both internal and external sources. That is, the organism does not have to absorb the radionuclide to suffer the exposure and the effect. The second mode of toxic effect is chemical toxicity, the same effect in a general sense as that resulting from most other contaminants. Chemical toxic effects imply absorption of the contaminant into the tissues. Of the several radionuclides released from nuclear facilities that have the potential to be chemically toxic, only U has the potential to be toxic as a result of releases from Canadian nuclear facilities. The two modes of toxic effect are handled separately in this assessment, with slightly different methods suitable to each. The potential combined effects of U and ionizing radiation were not accounted for in this assessment.

3.1 Pathways analysis

The behaviour and fate of radionuclides released from nuclear facilities to the environment are described in the supporting document (Bird *et al*., 2000). Uranium and Th and their decay chain daughter radionuclides are the radionuclides of concern released from U mines, whereas U is the main radionuclide of concern released from U refining and conversion facilities. Radionuclides of concern in TMAs are primarily ^{226}Ra and U, although other radionuclides may also be important in some WMFs located in Ontario (e.g., ^{3}H, ^{14}C, ^{60}Co, ^{90}Sr and ^{137}Cs). The environmental behaviour of U, radium (Ra) and other U and Th decay chain radionuclides has been extensively studied and the subject of numerous reviews (Sheppard, 1980; Gascoyne, 1992). A review of U chemistry and behaviour in the environment is provided by Macdonald (1999), while the environmental chemistry and behaviour of activation and fission radionuclides are reviewed by ECOMatters Inc. (1999a).

The major fission and activation products released from NGSs include ^{3}H, ^{14}C, ^{51}Cr, ^{54}Mn, ^{59}Fe, ^{60}Co, ^{65}Zn, ^{90}Sr, ^{95}Zr, ^{106}Ru, ^{124}Sb, $^{124–135}$I, ^{137}Cs and ^{144}Ce. Releases of radionuclides from these facilities are primarily to air or to water. Emissions to air will result in deposition of particle-reactive radionuclides and scavenging of radionuclides from the plume with distance from the source. Mobile radionuclides such as the inert gases will disperse quickly and reach background concentrations a short distance (a few kilometres) from the source. The dispersal of radionuclides deposited to soil or emitted to a surface water body is largely dependent on the hydrology of the system. Most of the radionuclides released are particle reactive and partition from the water to soil or sediment. A few radionuclides, such as ^{3}H, are very mobile. The bioavailability of the radionuclides and their uptake by biota ultimately govern their effect on biota.

Radiation doses to aquatic organisms are from external sources (e.g., radionuclides in water and/or sediment) and internal sources (e.g., radionuclides absorbed in tissues via respiration and ingestion). For terrestrial organisms, the radiation dose is from external sources (e.g., emersion in air and in soil) and internal sources (e.g., inhalation and ingestion for animals and absorption and respiration for plants). The internal radiation dose to organisms is estimated either directly from measured concentrations in the organisms or by the use of concentration ratios (CRs) to estimate the transfer of radionuclides from a given medium (e.g., soil, sediment or water) to the organism.

3.2 Assessment and measurement endpoints

Assessment endpoints are “a quantitative or quantifiable expression of the environmental value considered to be at risk in a risk assessment” (Suter, 1993). They are ecological features, often populations of biota, that are deemed important, exposed and sensitive to the nuclides of concern. The decision about importance is based primarily on quantitative evidence of ecological function. The assessment endpoints are related to the valued ecosystem components, but differ in that valued ecosystem components also account for stakeholder opinions about importance.

ENEVs for at least one assessment endpoint were developed for radiotoxicity and U toxicity in both the aquatic and terrestrial environments. In practice, only the most sensitive or most limiting assessment endpoint need be considered for a given medium. This is because if the primary or most sensitive assessment endpoint is protected, then the others are also protected. ENEVs were developed for most of the major taxonomic groups.

3.2.1 Uranium

Uranium is more chemotoxic than it is radiotoxic. For this reason, a separate assessment was performed to assess the toxicity of U to terrestrial and aquatic biota. Uranium mine releases of primary concern are those to waterways. Because macrophytes may have large CRs for radionuclides, including U (Bird and Schwartz, 1996), and muskrats (*Ondatra zibethica*) feed heavily on aquatic macrophytes, toxicity to the muskrat was selected as an endpoint for U toxicity to mammals. Uranium toxicity was also assessed for a dabbling duck, the mallard (*Anas platyrhynchos*), a fish-eating bird, the osprey (*Pandion haliaetus*), a fish-eating mammal, the mink (*Mustela vison*), and a carnivore, the red fox (*Vulpes fulva*), which is assumed to feed on small mammals (mice). Allometric equations are used to determine the metabolic energy needed by the animals. Exposure was estimated using the Canadian Wildlife Service (CWS) Wildlife Contaminant Exposure Model (WCEM), which uses empirical data and allometric relationships to estimate food and water intake in the exposed species. The endpoint to assess the potential effects of U chemical toxicity was kidney function of mammals and survival of rabbit. Chemotoxicity of U to crustacean zooplankton was assessed based on the U concentrations measured in the water column and data from both acute and chronic toxicity tests. Uranium toxicity to benthic invertebrates was assessed based on the U sediment concentration and the screening-level concentrations (SLCs) derived for northern Saskatchewan (Kurias *et al*., 2000), the location of Canada's operating U mines and U sediment toxicity data.

For the generic assessment of U chemotoxicity in the environment, endpoints included the reproductive success and population survival of:

- zooplankton
- benthic invertebrates
- fish
- waterfowl, the mallard
- muskrat
- osprey
- red fox
- terrestrial plants

3.2.2 Ionizing radiation

Measurement endpoints for this analysis relate primarily to species survival, productivity and reproduction (Section 3.4). Fish production was chosen as an assessment endpoint for exposure to radionuclides released to surface waters. Emphasis was on exposure of bottom-feeding fish, e.g., the white sucker (*Catostomus commersoni*) and brown bullhead (*Ictalurus nebulosus*), that tend to accumulate higher levels of radionuclides than do piscivorous species, such as lake trout (*Salvelinus namaycush*). Because of the high sediment/water partition coefficients (K_ds) of many radionuclides, sediment-dwelling macroinvertebrates are exposed to much higher levels of radionuclides than invertebrates and fish living in the water column. Therefore, reduction in the numbers of benthic invertebrates was chosen as a direct assessment endpoint for sediment exposure.

The most sensitive plant-related measurement endpoints evaluated were seed germination, seedling emergence and growth, growth at mid-phase and at maturity, and flower and seed production. Population survival of terrestrial plants was chosen as an assessment endpoint for exposure to radionuclides in the air or accumulated in the soil. Survival of soil invertebrates was an assessment endpoint for exposure of decomposers to radionuclides in the soil and litter. The most sensitive soil invertebrate-related measurement endpoints evaluated were survival, growth and production of offspring. Radionuclides may also accumulate in aquatic macrophytes and phytoplankton. Primary production (germination, growth and photosynthesis) was selected as an assessment endpoint for aquatic plants.

For the present assessment, appropriate assessment endpoints for radiotoxic effects could include fish species, phytoplankton (algae), benthic macroinvertebrates and macrophytes. For the generic assessment of radiotoxicity in the environment, endpoints included the reproductive success and population survival of:

- populations of fish
- populations of benthic invertebrates
- phytoplankton
- macrophytes
- small mammals (voles)
- soil invertebrates
- soil litter invertebrates
- terrestrial plants

3.3 Uranium

This section provides information on U toxicity that will be used to develop Critical Toxicity Values (CTVs) and ENEVs, as well as some information on environmental transport, speciation and biokinetics. The chemical toxicity of U to wildlife has been reviewed by Macdonald (1998).

Uranium is a member of the actinide series of elements and has an atomic number of 92. It has 10 radioactive isotopes, but ^{238}U (99.27%), ^{235}U (0.72%) and ^{234}U (0.0055%) are the three most common isotopes. The dominant isotope, ^{238}U, has a physical half-life of 4.5×10^9 years, giving it a very low specific activity (1.24×10^4 Bq·g^{-1} U). For these reasons, U is primarily a chemical toxicant and is not considered very radiotoxic. Because U has a high K_d value in most soils and a relatively low mean plant/soil CR of 0.0045, it has generally low bioavailability to plants. U does not biomagnify and transfers fairly inefficiently through the food web.

The transfer of compounds between the diet and the tissues of animals has traditionally been modelled using transfer coefficients (TCs) and CRs, even though ratios cannot be easily applied from one system or species to another (McGee *et al*., 1995). This approach has been used extensively for radionuclide exposure assessments in humans, but its use for ecological risk assessments is problematic, in part because of the scarcity of relevant TCs and CRs for non-agricultural species. A more reliable approach is to develop biokinetic parameters for the species used as an assessment endpoint, similar to the approach used in humans (Wrenn *et al*., 1995). The present assessment followed such an approach for the assessment of potential U toxicity to wildlife. Energy requirements for the five representative wildlife species assessed were calculated from allometric equations; in the absence of data on species-specific TCs, the TC values were calculated from the allometric equations (Macdonald, 1998). In most cases, U concentrations in the diet were based on measured values. However, when this information was not available, CRs were used to estimate the U concentration in the diet.

3.3.1 Effects characterization

3.3.1.1 Intake pathways and biokinetics

Biokinetics refers to the uptake, transport and distribution of U within an organism after ingestion. The amount of U that resides in an organ at steady state is proportional to the amount of ingested U absorbed from the gut, the fraction of absorbed U deposited in the organ and the biological half-time of U in that organ (Kocher, 1989). Almost all of the daily U intake is from food and water.

The distribution of U in wildlife in Canada is assumed to be similar to that in humans and measured in small mammals in chronic exposure studies. This suggests that the major site of U burden in wild species is the skeleton, followed by the kidney, which is considered to be the critical organ in terms of effects. However, at least one report on U concentrations in moose at Elliot Lake (MacLaren Plansearch Inc., 1987) showed that U was distributed fairly evenly in kidney, liver, heart and muscle. In one animal, very high concentrations were reported for the skin, ranging as high as 16.4 mg·kg^{-1}, which may indicate high dermal exposure.

Inhalation has been studied as a major source of U in humans and laboratory animals because of the potential for high exposures from wind-blown dust and locally contaminated atmosphere in industrial

settings. Similar exposure scenarios are possible for wildlife living near tailings, mill vents and U-rich areas. Inhalation is a major route of exposure that results in high U concentrations in target tissues, although the specific tissue involved depends on the U species. Inhalation of insoluble U results in accumulation in lung, tracheal/bronchial lymph nodes and spleen. Clearance from the lung is by mechanical action. Inhalation of soluble U results in accumulation in kidney and bone and clearance in urine.

Dermal exposure may be a significant pathway for a broad range of animal species in U-rich areas, including burrowing mammals and birds (e.g., burrowing owl [*Athene cunicularia*]). Yuile (1973) cites studies indicating that dermal exposure to U can cause severe U poisoning and death. Therefore, dermal exposure may be a major exposure route in some species.

Ingestion, which includes intake from food, water and soil, is probably the major exposure route for wildlife in Canada. A key consideration in determining the importance of ingestion is the assimilation rate from the gut — the absorption factor or f_1 parameter in metabolic models. This value determines how much material is ultimately transported to the blood for deposition in the critical organs. Ingestion rates have been reviewed by Wrenn *et al*. (1985, 1995), Durbin and Wrenn (1975), Yuile (1973), the ICRP (1979) and Leggett and Harrison (1995) for development of models to estimate allowable daily intakes in humans. These reviews provide data on laboratory studies with mammals that can be applied to feral species.

The uptake rate of U from the gut is generally low, ranging from <1% to 3–4% of the diet (Leggett and Harrison, 1995). Factors affecting this rate are the feeding status of the organism and the species (Morris and Meinhold, 1995). The ICRP (1979) has assumed a conservative absorption rate of 0.05 (5% of diet) for soluble U in adult humans and 0.002 (0.2%) for insoluble U. The difference in rates between soluble and insoluble forms is consistent with uptake rates in hamsters (0.77% for uranyl nitrate [$UO_2(NO3)_2 \cdot 6H_2O$]; 0.11% for UO_2) (Harrison and Stather, 1981). Wrenn *et al*. (1985) suggested an absorption rate of 1–2% for humans, which was modified slightly by Leggett and Harrison (1995) to 1–2.4% based on newer analyses. Slightly higher rates of uptake (up to 5%) are reported for water compared with food (Leggett and Harrison, 1995). The value of 5% is considered conservative for humans and at the top of the range of values reported for non-human species. A value of 5% was adopted as a guideline for wildlife and should account for the vast number of species for which there are no data.

Much higher U absorption rates may occur in neonates of some species. For example, U absorption is about 100 times higher in neonate rats than in adults. Absorption rates greater than 34.5% were reported for 2-day-old neonate swine (Sullivan, 1980).

Soil ingestion may represent a major source of U for wildlife (Beyer *et al*., 1994) but is generally not addressed in feeding studies. Linsalata *et al*. (1989) estimated that over 90% of the daily U intake in cattle from natural sources may be from soil. Animals may consume soil incidentally, such as

from the ingestion of roots and vegetation with deposited soil, or may intentionally ingest soil at lick sites that provide nutrients during the spring (Weeks and Kirkpatrick, 1976). Sheppard (1995) suggests a mean ingestion rate of 50 mg soil·kg^{-1} food in the diet of animals for modelling purposes; however, much higher rates have been reported for some feral species. Weeks and Kirkpatrick (1976) report inorganic fractions in deer feces as high as 87% in some individuals feeding at deer licks. High soil ingestion rates (30%) are also reported for shorebirds (Beyer *et al*., 1994).

The amount of U retained from dietary soil is a function of the total amount of U ingested and its bioavailability (Sheppard, 1995). At present, there is no reliable method to predict the bioavailability of U through this exposure route on a general scale. A gut absorption rate of 0.05 (5%) is used for adult mammals, with a higher value of 0.10 (10%) for juvenile and immature animals. A value of 30% is recommended for neonates until further data are available.

3.3.1.2 Terrestrial mammals and birds

Several major reviews on the specific mechanisms of chemical toxicity of U to mammals are available (Durbin and Wrenn, 1975; Leggett, 1989; Ribera *et al*., 1996). Although U accumulates in bone and kidney (Yuile, 1973; Durbin and Wrenn, 1975; Leggett, 1989), chemical effects have been observed in kidney only. Uranium toxicity is by the disruption of kidney function by binding to the membranes of renal tubular cells, restricting the reabsorption of glucose, sodium, amino acids, protein, water and other substances, and causing cell death by suppression of cell respiration (Leggett, 1989). Damage to kidney becomes evident as the cells of the proximal tubules stop functioning and die. Effects observed in bone (i.e., osteosarcoma) have been attributed to radiotoxicity and not chemical toxicity (Ribera *et al*., 1996). Reviews of U uptake indicate that birds retain greater amounts of U than mammals, given the same levels of exposure (Davis *et al*., 1993; Linsalata, 1994). However, until further evidence is available, it will be assumed that birds and mammals are of equal sensitivity to U exposures for this assessment.

For U, lethality is a common endpoint used for acute studies (Table 1), while kidney function is the predominant endpoint for chronic studies. Acute exposures to U have demonstrated that U is low in toxicity to most species, often requiring concentrations greater than 2% in diet to elicit an acute response. Rabbits are several times more sensitive than other species, with LD_{50} values below 0.5% of the diet. Insoluble U is generally non-toxic at concentrations up to 20% of the diet. An LD_{50} value of 23 mg U·kg^{-1} bw·d^{-1} has been reported for the rabbit in a 30-day feeding study with soluble U (Durbin and Wrenn, 1975). This value represents an acute toxicity value for the most sensitive species.

Considerable evidence is present in the literature to suggest that the threshold level of U in kidney may be a scientifically sound subchronic basis for setting the CTV. The International Commission on Radiological Protection (ICRP, 1959) adopted a maximum permissible concentration of 3 mg·kg^{-1} ww in kidney for the protection of humans, but reduced the value to 1 mg·kg^{-1} as indicators of kidney function became more sensitive (Wrenn *et al*., 1985; Bosshard *et al*., 1992). Morris and Meinhold

(1995) note that damage to kidney occurs in virtually all species at levels of 0.5 $mg \cdot kg^{-1}$ ww, consistent with the conclusions of other authors (Bosshard *et al*., 1992). They also note that although the kidney has a lot of reserve capacity and may function with the loss of a proportion of the nephrons, the loss of reserve capacity should be considered to be an adverse effect. Since animal studies have shown effects at kidney concentrations of 0.5–1.0 mg $U \cdot kg^{-1}$ ww, the value of 0.5 mg $U \cdot kg^{-1}$ ww in kidney should be considered to be a reasonable threshold level for the protection of most species.

Using the reported kidney threshold concentration of 0.5 mg $U \cdot kg^{-1}$ ww, a food:kidney TC of 2×10^{-3} $d \cdot kg^{-1}$ and other assumptions of nominal intakes of feed (Macdonald, 1998), the minimum dose to achieve the kidney threshold concentration is 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$. Given the weight of evidence for irreversible chronic effects at kidney tissue concentrations for most species tested, a dose of 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$ is recommended for the ENEV. This value is about equivalent to the use of the acute LD_{50} value for rabbits (23 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$) as a CTV with an application factor of 100 as recommended in Environment Canada (1997) to derive an ENEV of 0.23 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$.

3.3.1.3 Terrestrial plants

Uranium does not biomagnify up food chains and has relatively low toxicity to plants. For example, relatively high concentrations (>42.4 $mg \cdot L^{-1}$) are required before effects (germination and growth) are observed in plants grown in Hoagland's nutrient solution (Weinberger and Murthy, 1985). Tomato seedlings are more sensitive to U than other species tested, showing a 24% reduction in growth relative to controls at 4.24 mg $U \cdot L^{-1}$ and a 44% reduction at 42.4 mg $U \cdot L^{-1}$. In a literature review, Sheppard (1989) reported U toxicity to plants at soil concentrations ranging from 0.5 to 2600 $mg \cdot kg^{-1}$ dw. However, studies showing toxicity at or below 100 $mg \cdot kg^{-1}$ dw soil were not well documented or were impossible to confirm.

In garden soil (high organic carbon content), seed germination of corn, lettuce, tomato, rapeseed (*Brassica rapa*), and pine was not affected by U at concentrations as high as 1000 $mg \cdot kg^{-1}$ dw soil, presumably because of lower concentrations of bioavailable forms of U (Sheppard *et al*., 1992). In comparison, seed germination was significantly reduced at 1000 mg $U \cdot kg^{-1}$ dw soil applied to limed brunisol soil. Pine seeds exposed to 1000 mg $U \cdot kg^{-1}$ dw soil did not germinate, whereas seed germination was not affected by 300 mg $U \cdot kg^{-1}$ dw soil.

Sheppard *et al*. (1989) report a K_d value of 15 $L \cdot kg^{-1}$ for the most U-contaminated soil that they tested from Port Hope, Ontario. Using this K_d value and a concentration of 4.24 mg $U \cdot L^{-1}$ from Weinberger and Murthy (1985) gives a soil U concentration of about 64 $mg \cdot kg^{-1}$ for reduction in growth of tomatoes. This value was chosen as the CTV. Using an application factor of 1 gives a realistic ENEV of 64 mg $U \cdot kg^{-1}$ dw soil. This value is above the maximum U concentration of about 11 $mg \cdot kg^{-1}$ reported by Kabata-Pendias and Pendias (1992) in uncontaminated surface soils from the United States and Canada.

3.3.1.4 Soil invertebrates

In garden soil, survival of earthworms (*Lumbricus terrestris*) to 75 days was not affected by U concentrations of 3–1000 mg·kg^{-1} dw soil (Sheppard *et al*., 1992). Using the No-Observed-Effect Concentration (NOEC) of 1000 mg·kg^{-1} dw soil for earthworms as the CTV and an application factor of 10 gives an ENEV of 100 mg·kg^{-1} dw soil. Again, this value is well above the maximum U concentration reported in uncontaminated surface soils from the United States and Canada (Kabata-Pendias and Pendias, 1992).

3.3.1.5 Pelagic aquatic organisms

Uranium does not bioaccumulate to a great extent, although tissue:water and tissue:food CRs greater than one are common. There is also no evidence that U biomagnifies through the food web, probably because of its very low rate of uptake (i.e., <5%) through the gut of most organisms. As a result, the concentration of U in upper trophic levels is often much lower than in the bottom trophic levels. The lower trophic levels that accumulate U from inorganic matrices (i.e., soil or sediment) tend to be more highly exposed than upper trophic level species.

Acute and chronic toxicity data for fish and invertebrate species found in Canada are summarized in Tables 2 and 3. Older fish are about two times more tolerant of U under acute exposure conditions (Holdway, 1992). Holdway (1992) calculated a threshold toxicity for growth in a larval tropical fish of 0.20 mg·L^{-1} in soft water. Bywater *et al*. (1991) reported U LC_{50} values ranging from 0.73 to 3.46 mg·L^{-1} for six tropical species of fish and from 0.14 to 0.9 mg·L^{-1} for four species of cladocerans in water low in hardness and alkalinity. In contrast, fish species tested in water at higher hardness and alkalinity had LC_{50} values several times higher. Tarzwell and Henderson (1960) reported LC_{50}s for fathead minnow (*Pimephales promelas*) of 2.8–3.1 mg·L^{-1} in water with hardness of 20 mg·L^{-1} and an LC_{50} of 135 mg·L^{-1} in 400 mg·L^{-1} hardness.

For this assessment, the 96-h LC_{50} for fathead minnow of 2.8 mg·L^{-1} (Tarzwell and Henderson, 1960) will be used for the CTV for fish. An application factor of 10 is applied to the CTV to estimate a chronic effect level of 280 µg·L^{-1}.

There is a strong correlation between water hardness and U toxicity to aquatic organisms. The hardness in the lakes in the Beaverlodge, Saskatchewan, area is approximately 36–80 mg·L^{-1} (Macdonald, 1998). Data from other monitoring sites near Cigar Lake, Saskatchewan, indicate a water hardness of <10 mg·L^{-1}. This is the level of hardness at which U is most toxic. In general, the hardness of water is low (<10 mg·L^{-1}) in Canadian Shield lakes near U mines and mills but may be elevated (>100 to >1000 mg·L^{-1}) in those waters receiving treated mine effluent discharges from water treatment plants.

Pickett *et al*. (1993) derived a very low mean LC_{50} of 3 $\mu g \cdot L^{-1}$ when testing the effect of U on reproduction in *Ceriodaphnia dubia* in soft water. This study is notable in that the reported LC_{50} is the mean of studies conducted in two independent laboratories and hence is reproducible. The CTV and ENEV (assuming an application factor of 1) for *Ceriodaphnia* in soft water is 3 $\mu g \cdot L^{-1}$. Poston *et al*. (1984) reported a Lowest-Observed-Adverse-Effect Level (LOAEL) of 520 $\mu g \cdot L^{-1}$ for daphnid reproduction in water with a hardness of 66–73 $mg \cdot L^{-1}$ (similar to the Beaverlodge Lake region). Greater sensitivity was reported in a study by Trapp (1986), who reported a 48-h LC_{50} of 220 $\mu g \cdot L^{-1}$ for *Daphnia pulex* in soft water. Using an acute:chronic ratio of 10 (Poston *et al*., 1984), this would give a chronic toxicity value of 22 $\mu g \cdot L^{-1}$. This value was chosen as the CTV. Using an application factor of 1 because of the high sensitivity of the study compared with others would give an ENEV of 22 $\mu g \cdot L^{-1}$ for *Daphnia* (zooplankton). This value is greater than the 95th percentile of 0.35 $\mu g \cdot L^{-1}$ for background U concentrations in Saskatchewan and 0.28 $\mu g \cdot L^{-1}$ in Ontario (Friske, 2000).

3.3.1.6 Benthic invertebrates (benthos)

Sediments are a sink for particle-reactive contaminants released into aquatic ecosystems. Sediment-associated contaminants may have adverse effects on benthic macroinvertebrates and fish that feed on contaminated invertebrates. There are few or no published data on the toxicity of sediment-bound U to benthic invertebrates. To our knowledge, the only study to investigate the toxicity of sediment-bound U to benthic invertebrates is a recent study by BEAK International Inc. (1998). BEAK International Inc. (1998) exposed both juvenile and adult *Hyalella azteca* to U-spiked sediment. Juvenile amphipods were much more sensitive to U than the adults. From their study, an LC_{20} of 15 mg $U \cdot kg^{-1}$ dw sediment and an LC_{50} of 57 mg $U \cdot kg^{-1}$ dw sediment were calculated for juvenile *Hyalella azteca*.

Because no guidelines are available for U in sediment, Kurias *et al*. (2000) developed regional-specific SLCs for northern Saskatchewan, the location of Canada's operating U mines. They used environmental monitoring data for sediment contaminant concentrations and co-occurring benthic invertebrate monitoring data in northern Saskatchewan lakes near operational and pre-operational U mine sites and the SLC approach (Neff *et al*., 1986) to calculate the Lowest Effect Level (LEL) and Severe Effect Level (SEL). The LEL corresponds to the value at which actual ecotoxic effects become apparent. The SEL corresponds to the value that could potentially eliminate most of the benthic organisms (Persaud *et al*., 1992).

The SLC approach is an effects-based approach applicable to benthic organisms and is an estimate of the highest concentration of a contaminant that can be tolerated by a specific proportion of the benthic species; it may capture the combined effects of simultaneous exposure to several contaminants that may be present in an area. The LEL calculated for U in sediment was 21 $mg \cdot kg^{-1}$ dw sediment, and the SEL was 390 $mg \cdot kg^{-1}$ dw. These values are a best estimate of the potential toxicity of sediment-bound U to benthic invertebrates and were derived following similar procedures used by the Ontario Ministry of the Environment to derive their sediment quality guidelines for other contaminants (Persaud *et al*., 1992).

On the basis of the above, the CTV for sediment-bound U is in the range of 15–21 $mg \cdot kg^{-1}$ dw sediment. The value of 21 $mg \cdot kg^{-1}$ dw sediment was chosen as both the CTV and the ENEV. This value is close to the 95% confidence limit (CL) of 29.5 $mg \cdot kg^{-1}$ dw sediment (the 90% CL is 17 $mg \cdot kg^{-1}$ dw sediment) for background sediment U concentrations in northern Saskatchewan (Friske, 2000). A lower CTV would be inappropriate because it would be well in the range of normal background concentrations. Even with a CTV near the 95th percentile of background, very little additional contamination would cause an exceedance of the ENEV in some areas. The CTVs and ENEVs for wildlife and aquatic biotic used in this assessment are summarized in Table 4.

3.3.2 Exposure characterization and risk analysis

This section describes the exposure values estimated and the risk quotients calculated to assess the effects of U exposure on wildlife and aquatic species in the Canadian environment. The EEV for representative species is calculated and compared with the ENEVs for wildlife and other terrestrial and aquatic biota (Macdonald, 1998). Wildlife diet information and food and water intake rates were calculated using the WCEM of the CWS (Brownlee, 1999) using methods derived from the *Wildlife Exposure Factors Handbook* of the U.S. Environmental Protection Agency (U.S. EPA, 1993). Empirical data and allometric relationships were used to estimate food and water intake in the exposed species. Exposure to U was calculated by combining the food and water intake rates with environmental levels of U near nuclear facilities. Uranium concentrations used to establish EEVs are the most recent data available, typically representing the years 1995–1998. Nuclear generating stations were not considered as a source of U exposure in the following sections because they do not release U in detectable quantities.

3.3.2.1 Biokinetic method used for wildlife

The aquatic system is the major system expected to be affected by the release of U from mines, mills and other nuclear facilities. Wildlife feeding in the affected waterway can ingest very high levels of U from water and other environmental media. For this reason, an accurate mean concentration of waterborne U for both background areas and impacted areas is crucial for a realistic risk assessment. Mean water concentrations are used in this assessment to estimate the amount of U taken in daily by the representative species and to directly estimate toxicity of U to aquatic organisms. In northern Saskatchewan, the background median aqueous U concentration is below the detection limit of 0.05 $\mu g \cdot L^{-1}$, with a 95th percentile of 0.35 $\mu g \cdot L^{-1}$. The median sediment concentration is 3.7 $mg \cdot kg^{-1}$ dw sediment, with a 95% CL of 20.5 $mg \cdot kg^{-1}$ dw sediment. In Ontario, the background median aqueous U concentration is <0.05 $\mu g \cdot L^{-1}$, with a 95th percentile of 0.28 $\mu g \cdot L^{-1}$. The median sediment concentration is 4.2 $mg \cdot kg^{-1}$ dw, with a 95th percentile of 51 $mg \cdot kg^{-1}$ dw (Friske, 2000).

Risk quotients were estimated based on the levels of exposure in the red fox, mink, muskrat, mallard and osprey. The species were selected because of their broad distribution in Canada and hence a strong likelihood that they could be exposed to U discharged from a mine, mill or TMA. The assessment was conducted in the initial stages using the WCEM of the CWS. The model calculates the amount of food and water consumed on a daily basis from literature values or allometric equations. Soil/sediment intake was calculated separately as a function of the amount of mass ingested by the organisms daily. These values (Table 5) are used with environmental concentration data to derive the levels of intake. In addition, their placement in the food web is such that they would be maximally exposed if the U accumulated in their food species. The items in the diet of most of the species have been adjusted to give maximum exposure through a specific food type (e.g., 100% aquatic plants for muskrat, 100% fish for osprey and mink, or 100% small mammals for fox). Soil/sediment intake was estimated from the literature (Beyer *et al.*, 1994), with a default value of 2% of the diet used for species, like the osprey, that are unlikely to be exposed to large amounts of soil/sediment. Higher ingestion rates (5%) were assumed for some species, such as those that burrow in soil or mud and are likely to be exposed to higher soil intake levels.

Total daily intake (TDI) of U was calculated by:

$$TDI\ (\mu g \cdot d^{-1}) = I_{water} + I_{food} + I_{soil/sediment}$$

where:

- TDI = total daily intake ($\mu g \cdot d^{-1}$),
- I_{water} = intake from water ($\mu g \cdot d^{-1}$),
- I_{food} = intake from food ($\mu g \cdot d^{-1}$), and
- $I_{soil/sediment}$ = intake from soil/sediment ($\mu g \cdot d^{-1}$).

The daily mass in each intake pathway was estimated using empirically based allometric equations that use body weight (bw) in grams to estimate the physiological parameters (Peters, 1983; Sample *et al*., 1997). Equations and assumptions for U intake by wildlife are described in Macdonald (1998). Exposure to U by drinking water is estimated from the amount of water required by the animals for metabolic activity and reported aqueous U concentrations. Likewise, intake of U in food was estimated from the energy requirements of the animals, the energy content of the food items and the U concentration in the food. Energy requirements were derived from the allometric equations. The caloric contents of food items are taken from published reports.

Most of the data are taken from species accounts in Burt and Grossenheider (1976), Banfield (1974) and Sample and Suter (1994). Body weights, water intake rates, food intake rates and soil/sediment ingestion rates are given in Table 5. Sample and Suter (1994) report a diet that consisted of 68.8% mammals, 12% birds, 10.4% plants, 0.9% insects and 5.5% other items for the red fox and 46% mammals, 16% fish, 15% aquatic invertebrates, 13% amphibians and 8% birds for the mink. A large proportion of the muskrat's diet is aquatic vegetation (macrophytes). The mallard is a dabbler that

feeds by “tipping up” in shallow water and sifting seeds, plants and invertebrates out of sediments (Terres, 1982). The osprey is a predatory bird that eats virtually only fish, although small amounts of snakes, frogs and other birds may also be eaten (Terres, 1982).

Data given in Table 5 for body weight for the five wildlife species and their ingestion of water, food and soil, based on their metabolic energy demands, were used to compute EEVs based on reported U concentrations in water, sediment, fish, macrophytes and aquatic invertebrates. Where data were not available for concentrations in food components of the diet (fish, macrophytes and aquatic invertebrates), U concentrations in biota were estimated using geometric mean (GM) values for CRs reported in Bird and Schwartz (1996), except for macrophytes, where the region-specific CRs, and hence more realistic CRs, were used. All concentrations in aquatic invertebrates were estimated using the GM CR. In a few instances, U concentrations are reported in sediment but not water, or in water but not sediment. Risk quotients in these instances are flagged in tables in which this occurs to indicate that either the sediment or water pathways are missing in their calculation.

3.3.2.2 Uranium mines and mills

There are currently no operating U mines in Ontario and three fully operating U mines and mills in northern Saskatchewan. They are the Rabbit Lake and Key Lake sites operated by Cameco Resources Inc. and the Cluff Lake site operated by Cogema Resources Inc.

3.3.2.2.1 Elliot Lake region mines

Following the discovery of U deposits in the mid-1950s, 11 mines and 10 associated mills were opened within the Serpent River watershed in the region surrounding Elliot Lake. Rio Algom Ltd. and Denison Mines Ltd. were responsible for all mining operations by the mid-1960s. Prior to the 1970s, mine wastes and effluents were released to the surrounding environment virtually untreated. Radium removal and pH control practices were initiated in the early 1970s.

Seven major WMFs or TMAs are located within the Serpent River watershed, including Denison (discharges to Quirke Lake), Quirke (discharges to Quirke Lake), Panel (discharges to Quirke Lake), Stanleigh (treated effluent discharged to McCabe Lake), Stanrock/Can-Met (treated effluent discharged to Moose Lake), Lacnor/Nordic (treated effluent discharged to Nordic Lake) and Spanish-American (discharges to Denison TMA #1).

The main branch of the Serpent River originates in the northwest end of the watershed at Ten Mile Lake and drains southward to Dunlop Lake, flows east through Quirke Lake, enters Whiskey Lake from the northwest and leaves through the southwest. The Serpent River then flows west into Pecors Lake and southward to McCarthy, Shedden and Camp lakes, then west to Serpent Harbour and Lake Huron. Another branch of the river flows east and south through McCabe, May and Hough lakes into Pecors Lake. The midwestern portion of the watershed drains through the Marshland River,

starting above Gullbeak and Elliot lakes, flowing eastward through Esten, Marshland, Grandeur, Trout and Depot lakes into McCarthy Lake.

Dunlop Lake, upstream of the TMAs/WMFs, provides an upstream reference station and is considered to represent pre-mining baseline conditions. Quirke Lake is the largest lake in the Serpent River watershed. Quirke Lake receives treated effluent from the Denison TMAs and Quirke and Panel WMFs. Pecors Lake receives water from Whiskey Lake (downstream of Quirke Lake) and from the McCabe–May–Hough chain of lakes. Water quality of the McCabe–May–Hough chain is affected by treated effluent from the Stanrock TMA, which flows into May Lake. McCarthy Lake is downstream of Pecors and also receives flow from the Elliot Lake chain (Elliot–Quimby–Esten–Marshland–Grandeur–Trout–Depot). Treated effluent from the Nordic WMF enters Nordic Lake and then the Esten Lake diversion to the Elliot Lake chain. The Serpent River at Highway 17 receives flow from 97% of the Serpent River basin, and the water quality at this station may be considered representative of the combined flow from the entire watershed. The Marcellus Creek watershed contributes the remaining 3% of the inflow, but is not impacted by mining activities.

Several radionuclides are monitored in water, sediment and biota in the Serpent River watershed. Elevated radionuclide concentrations are evident in areas receiving effluent from the TMAs/WMFs in comparison to reference locations.

Realistic RQs for the Serpent River watershed indicate that U is potentially toxic to biota (i.e., RQs >1) in several water bodies (Table 6). In particular, high risk quotients are calculated for McCabe Lake, which receives treated effluent from the Nordic WMF. Potential U toxicity to wildlife is predominantly from consumption of contaminated fish. This is illustrated by the relatively large number of RQs greater than 1 for osprey and mink. Sediment is another important pathway for potential U toxicity, with RQs greater than 1 for benthic invertebrates in several lakes. Uranium concentrations in water tend to be low and pose a potential risk only to the most sensitive pelagic species, *Ceriodaphnia*. Risk quotients greater than 1 for osprey and mink in Dunlop Lake, a headwater reference lake, reflect the relatively high U concentrations measured in fish from this lake, whereas the RQ greater than 1 for benthic invertebrates reflects a relatively high background sediment U concentration.

3.3.2.2.2 Beaverlodge Lake, Saskatchewan

A mill was built and put into operation near Ace Creek below Ace Lake in 1952 to mine local ore deposits. Satellite mine sites that included both underground and surface mining were in operation between 1970 and 1982. A total of 12 open pit mines were established over the 30-year period of production at Beaverlodge Lake. The last ore was hoisted from the Fay mine in June 1982, and the mill ceased processing in August 1982. An alkaline leaching process was used to extract the U, with the recovery ranging from 84% to 97%.

The tailings management system consisted of two natural lakes, Fookes and Marie lakes, for tailings solid settling. An artificial lake, Meadow Lake, was used to settle out particulate and precipitated Ra following the addition of barium chloride ($BaCl_2 \cdot 2H_2O$) at the Marie Lake treatment plant. Fookes Lake was the primary surface tailings disposal location, receiving 6 million tonnes. Marie Lake received 170 000 tonnes, and 101 000 tonnes were placed in Minewater Lake. The remainder of the tailings (40%) was placed underground as mine backfill. A tailings beach developed at Fookes Lake and at Marie Lake. Minewater Lake, which naturally flowed into Ace Creek, was diverted into the Fulton Creek system.

The Beaverlodge mining area has been decommissioned. Post-decommissioning activities mainly involve monitoring of the various sites. Close-out objectives for this decommissioned mining area are the Saskatchewan water quality objectives of 250 $\mu g \cdot L^{-1}$ for U and 0.11 $Bq \cdot L^{-1}$ for ^{26}Ra. Current U releases to the water column are from WMFs and from sediments.

Uranium concentrations in water and sediment along with realistic RQs for wildlife and aquatic biota are presented in Table 7. Note that there are few recent measurements of sediment U concentrations. Most of the data presented in Table 7 are from the late 1970s and early 1980s. Only a few sites have been sampled since decommissioning in the mid-1980s. Also, sediment concentrations tend to represent the top several centimetres of sediment (10–15 cm) and therefore do not necessarily represent the quality of recently deposited sediment, which composes the top centimetre or so of sediment or the top few centimetres of bioturbed sediment.

Risk quotients were greater than 1 at most locations, the exception being Ace Creek (monitoring station AC-14) upstream from Beaverlodge Lake. Even here, aqueous U concentrations were high and potentially toxic to pelagic species. The broad distribution of RQs greater than 1 for both wildlife and aquatic biota at most sites illustrates that water, sediment and consumption of macrophytes and fish are all important pathways for the potential toxicity of U to biota in these aquatic systems.

3.3.2.2.3 Cluff Lake mine, Saskatchewan

The Cluff Lake mine is located in the Island Creek watershed. The drainage system includes Snake Lake, Snake Creek, Island Lake, the fen/wetland area west of Island Lake, Agnes Lake and Island Creek. Phase I of the Cluff Lake Project (1980–1983) involved open pit mining of the D ore body and subsequent processing of high- and medium-grade ore. The mill tailings from Phase I were processed during the Phase I extension between May 1983 and August 1984. Approval of Phase II of the operation was granted in June 1983 and involved mining of the Claude, N, O-P and Dominique-Peter ore bodies. In June 1994, open pit mining was initiated at the Dominique-Janine north pit, and mining of the Dominique-Janine extension (on the shore of Cluff Lake) is ongoing.

Water collected from the TMA is passed through a secondary treatment system before being discharged to a point immediately downstream of Snake Lake. Effluent from the TMA enters Island Lake, which is drained by Island Creek and flows into Sandy Lake before connecting with the Douglas River.

Radionuclide concentrations are available for water, sediment, fish, aquatic macrophytes, small mammals, lichen and soil in the impacted lakes and streams of the Island Creek watershed. Environmental monitoring data are available from the late 1970s (pre-operational) to the present.

Effluent discharge into Island Lake has resulted in an increase in aqueous and sediment U at several locations relative to baseline conditions. Concentrations of all measured radionuclides have generally remained unchanged in lake whitefish (*Coregonus clupeaformis*) and northern pike (*Esox lucius*) in Sandy Lake, whereas concentrations of U have increased by a factor of 3–4 in lake whitefish from Cluff Lake and in northern pike from Island Lake. Uranium concentrations have also increased in yellow pond lily (*Nuphar* sp.) in Island Lake. Uranium concentrations in soil and lichen near the mill are elevated compared with those at a reference site 11.5 km from the mill. Based on limited data, the mine has not resulted in an increase in tissue burdens of U, Th, ^{226}Ra or ^{210}Pb in small mammals (voles).

Uranium concentrations in aquatic media and realistic RQs for wildlife and aquatic biota are presented in Table 8. Risk quotients were greater than 1, particularly for the Island Lake outlet site. This is the first lake to receive effluent from the TMA, which is passed through a secondary treatment system. High U concentrations in water, sediment, fish and macrophytes indicate that these are all important pathways for the potential toxicity of U to wildlife and aquatic biota in Island Lake.

3.3.2.2.4 Rabbit Lake, Saskatchewan

Surface water quality in Wollaston Lake has been monitored since the start of operations in 1975. Sampling locations in the lake include Pow Wow Bay, Hidden Bay, Collins Bay, Ivison Bay and Wollaston Lake proper. In addition, Collins Creek, Harrison Lake and the Umpherville River are also monitored.

Diversion of effluents to the Link Lakes resulted in a maximum U concentration in sediment of 4000 mg·kg^{-1} dw at the Sedimentation Dam monitoring station S1 in 1984. The highest concentrations of U in sediment measured in downstream lakes were in Horseshoe Lake, where, for the period 1985–1997, concentrations varied between 980 and 2040 mg·kg^{-1} dw. At other stations, U sediment concentrations ranged between 4.4 and 430 mg·kg^{-1} dw, and the majority of the concentrations were below 25 mg·kg^{-1} dw.

Radionuclide concentrations are available for two species of fish, longnose sucker (*Catostomus catostomus*) and northern pike, in the Rabbit Lake Project area. Radionuclide concentrations in the aquatic macrophytes in the Rabbit Lake Project for 1979–1985 indicate no obvious temporal trends.

Concentrations of U, ^{230}Th and ^{226}Ra in lichen are highest at Site 2, the closest monitoring station to the Rabbit Lake Pit, and there is little variation in concentrations of ^{210}Pb among sites.

Uranium concentrations in aquatic media and realistic RQs for both wildlife and aquatic biota are presented in Table 9. Note that RQs for most of these sites are based only on U concentrations in sediment. Risk quotients are much greater than 1 for the Link Lakes and Horseshoe Lake. They are also greater than 1 for the regional basin maximum for sediment U concentrations, a baseline value considerably greater than the 98th percentile of 59 $mg \cdot kg^{-1}$ dw for U in sediment in Saskatchewan (Friske, 2000). Therefore, elevated natural pre-development sediment U concentrations in some of these lakes have the potential to cause effects. Lower RQs (<1) are generally recorded for Pow Wow Bay, Hidden Bay and Collins Bay Eagle Point.

3.3.2.2.5 Key Lake, Saskatchewan

The David Creek watershed is impacted by treated mill effluent, whereas the Outlet Creek watershed is affected by dewatering activities. Both David Creek and Outlet Creek flow into the Wheeler River and eventually to Russel Lake. Treated mill effluent is released to Wolf Lake, which subsequently discharges to Fox Lake, Yak Creek, David Creek, Delta Lake, Wheeler River and Russel Lake.

Groundwater collected as a result of dewatering activities at the Deilmann and Gaertner ore bodies is discharged to Horsefly Lake, which flows into Little McDonald Lake, McDonald Lake, McDonald Creek and finally to Outlet Creek and to the Wheeler River to Russel Lake.

Data on concentrations of U and other radionuclides are available for water, sediment, fish, macrophytes, lichen, vegetation, soil and small mammals from various monitoring stations in the Key Lake area. For the David Creek watershed, receiving treated mill effluent, surface water and sediment U concentrations are elevated relative to 1977–1978 baseline conditions, whereas concentrations of other radionuclides remained relatively unchanged.

In the case of fish captured in Delta and Russel lakes, there appears to be little change in concentrations of U and other radionuclides over time. In McDonald Lake, however, concentrations of U in lake whitefish tissue have increased over time relative to the 1977 baseline conditions. Concentrations of ^{210}Pb, ^{210}Po, ^{226}Ra and U have also increased in macrophytes.

Little variation was observed in soil radionuclide concentrations among sites, and temporal trends are not evident over the 1982–1997 period. However, concentrations are elevated in soil and small animals in areas affected by the above-ground tailings management facility. A temporal trend of increasing concentrations over time was evident for U in lichen and blueberry at the Douglas Lake sites, but is not always apparent in the case of the other radionuclides. In lichen, the highest U concentrations were measured at the Hourglass Lake and Marmot Lake stations, where concentrations varied from 3.7 to 10.6 $mg \cdot kg^{-1}$ dw.

Uranium concentrations in aquatic media, soil and small mammals along with realistic RQs are given in Table 10. Risk quotients generally were greater than 1 for Horsefly Lake and Little McDonald Lake and to a lesser extent McDonald Lake, indicating potential toxic effects on wildlife and benthic invertebrates. Horsefly Lake receives groundwater discharge (dewatering effluents) from the Deilmann and Gaertner ore bodies. Horsefly Lake drains into Little McDonald Lake then McDonald Lake, which accounts for the elevated sediment U concentrations in these two latter lakes. Note that the RQs for both Horsefly and Little McDonald lakes are based on the sediment pathway; data were not available for U concentrations in water, fish or macrophytes. Also note that the sediment U concentration of 212 $mg \cdot kg^{-1}$ dw in David Creek in 1992 may be an anomaly, as values were 3.6 $mg \cdot kg^{-1}$ dw in 1994 and 5.7 $mg \cdot kg^{-1}$ dw in 1997 and were also low in previous years. In general, RQs were less than 1 for Delta and David Creek and for the maximum reported baseline concentrations. The latter indicate that baseline U concentrations do not have the potential to cause harmful effects except to the most sensitive species, such as *Ceriodaphnia*.

3.3.2.2.6 Baseline conditions — Pre-operational mines, Saskatchewan

Maximum concentrations of U and other radionuclides in water, sediment, soil and biota (aquatic and terrestrial) from pre-operation data for Cigar Lake, McClean Lake, Midwest Joint Venture and McArthur River project areas were used to assess the potential toxicity of U and other radionuclides at baseline concentrations to biota in order to verify the suitability of the ENEVs selected for this assessment.

The Cigar Lake Project involves underground mining of U ore and disposal of mine rock at the Cigar Lake site and transporting a thickened ore slurry for milling at an off-site mill for final processing into yellowcake. The Cigar Lake watershed includes Seru and Longyear bays of Waterbury Lake and Aline, Cat, Dragon, Bizarre and Abysmal lakes. Final effluent discharge of the Cigar Lake mine water treatment plant flows south into a muskeg area and is released through a muskeg dike to Aline Lake, which discharges via Aline Creek 3 km north into Seru Bay of Waterbury Lake (Cigar Lake Mining Corporation, 1998). Because of its close proximity to McClean Lake, many of the baseline environmental monitoring data for the Cigar Lake Project come from the neighbouring McClean Lake facility.

The McClean Lake property is located in northern Saskatchewan about 350 km north of the town of La Ronge and 12 km northwest of a U mine operated by Cameco at Collins Bay. The property covers an area of 26 604 ha. The mine is expected to have an operational life of about 16 years. The mill will have a nominal U production capacity of 1800 tonnes per year. Mill process waste streams, collected runoff and tailings seepage water will be treated in a water treatment plant located at the mill. The treated water is pumped to Sink Lake for regulated discharge through Vulture Lake to McClean Lake. Kewen Lake, located downstream from McClean Lake, is the receiving water body in the far field. Mallen Lake, Collins Creek upstream of Nadai Lake, and the western basin of McClean Lake are

reference sites because of their isolation from effluent and air contaminant exposure. In addition, water quality is monitored in Fox, Pat, Johannes, Tail, Isis and Osiris lakes due to their proximity to the JEB Pit and potential exposure to Rn gas.

The Cogema McClean Lake Operation has been in construction and operational status. The construction of the mill and support facilities is complete, and open pit mining and stockpiling of ore from the JEB Pit are complete. The conversion of the JEB Pit to a tailings disposal facility and the licensing of the mill to operate were approved in 1999.

Midwest Joint Venture is a U mine proposed for development in northern Saskatchewan 30 km west of Wollaston Lake, near McClean Lake. Ore from the Midwest Mine is to be processed at the McClean Lake mill.

The McArthur River Project is located within the Athabaska region of northern Saskatchewan about halfway between the Key Lake and Rabbit Lake mines in the Close and Yalowega lakes area. The property covers an area of 83 099 ha. The project area lies within the May Creek drainage area, which drains through May Creek into Yalowega Lake, through the Baxter–Whitford system into Waterbury Lake. Water bodies in the area are categorized as exposure (Ore Zone Pond, Toby Lake, Boomerang Lake, Read Creek, Unnamed Lake #1, Little Yalowega Lake and Yalowega Lake) or reference (Read Lake, Lower Read Lake, Upper Read Lake and Boomerang Lake west basin) (TAEM, 1997). Work on the McArthur River Project has been ongoing since the late 1970s, with extensive exploration work leading to the discovery of the P2 North zone in 1988. Subsequent work has delineated this zone. The ore is located near Ore Zone Pond at a depth of 500 m below the surface and therefore requires construction of a shaft to extract the ore and supporting infrastructure (McArthur River Joint Venture, 1992). In August 1997, a construction licence was granted to the McArthur River Project to construct all necessary surface facilities and support infrastructures and to carry out an underground development and construction program, including the siting of No. 2 shaft. A mining operation licence was issued in 1999.

Maximum baseline U concentrations in aquatic media, soil and small mammals and realistic RQs at the pre-operational mines are presented in Table 11. Risk quotients for wildlife and aquatic biota were less than 1, with the exception of *Ceriodaphnia*, a sensitive pelagic zooplankton species, and benthic invertebrates in Nicholson Bay. In the case of benthic invertebrates, only 2 of 59 sites examined had an RQ greater than 1. Therefore, the U concentrations at baseline conditions tend not to have the potential to cause harmful effects and indicate that the ENEVs selected to assess the potential toxicity of U at operational U mines and mills are not overly conservative.

3.3.2.3 Uranium refineries and conversion facilities and waste management facilities

Uranium concentrate or yellowcake from the mills is upgraded by refining and conversion to UO_3 and then to UO_2 and UF_6. The UO_2 is used directly in the manufacture of fuel bundles for CANDU-type reactors, whereas the UF_6 is used as feed material for the U enrichment process. The enrichment process does not occur in Canada. Cameco carries out the refining and conversion processes in its facilities in Blind River and Port Hope, Ontario. At Blind River, yellowcake is made into UO_3. The UO_3 from Blind River is shipped to the Port Hope conversion facility, where the UO_3 is converted into UO_2 for domestic reactor fuel production and to UF_6 for export. Uranium metal is also produced in a Speciality Metal Plant at Port Hope (AECB, 1998).

In Canada, waste management facilities for radioactive wastes have developed to handle a variety of wastes, including contaminated soils (historical contamination), liquid and solid wastes, contaminated filters and resins from pollution control systems of nuclear power plants, and reactor spent fuel. Some facilities are well-engineered recent structures, while others are older, less well designed sites.

3.3.2.3.1 Blind River, Ontario

Blind River is the site of a U refinery owned by Cameco. The facility was originally owned and operated by Eldorado Resources Ltd. and began operations in 1983.

In 1981, the Ontario Ministry of the Environment initiated a program to obtain background data on soils and vegetation in the vicinity of the Eldorado Resources Ltd. refinery, prior to the commencement of operations. In 1982, a high-volume air sampler was installed on the roof of the St. Joseph's Hospital in Blind River. In 1992, the hospital air quality monitoring (high-volume) site was abandoned when the hospital was demolished. In 1993, an alternative site in the town of Blind River was put into service. Filters from this site are consistently analysed for radionuclide levels in suspended particulate matter to obtain representative air quality samples in the community. Since at least 1992, air quality has been monitored using high-volume air samplers at four sites in the vicinity of the refinery. These are operated continuously. Air filters are removed and analysed every 14 days.

A network of vegetation monitoring plots was established in the vicinity of Eldorado Resources Ltd. in 1981. At seven sites, 20 trees each of trembling aspen (*Populus tremuloides*) and eastern white pine (*Pinus strobus*) were evaluated annually between 1981 and 1987 for growth conditions and foliar concentrations of U. Soil and grass samples were also collected at each site over the monitoring period. A separate collection of foliage of other plant species was made in 1985, and wild edible plants (e.g., blueberry) were collected in 1986 in the vicinity of the refinery.

Radionuclide concentrations are also monitored in the Mississauga River, both upstream and downstream of the Blind River refinery. Operation of the refinery has had little effect on radionuclide concentrations.

On the basis of the limited data for U concentrations in water and soil, RQs for both wildlife and aquatic biota are less than 1 (Table 12), indicating little potential for effects.

3.3.2.3.2 Port Hope, Ontario

Eldorado Gold Mines started operating a Ra refinery in Port Hope in 1932 to recover Ra from pitch-blend ore mined at Port Radium, Northwest Territories. Eldorado Gold Mines was taken over by the government in 1944 and renamed Eldorado Mining and Refining Ltd. Radium residues produced at the plant in the early 1930s were disposed on-site until 1939. In the early 1940s, the plant switched its operation from Ra extraction to the production of U. The Ra laboratories were dismantled in 1955–56 and buried at the Welcome Residue Area. Wastes produced at the refinery were disposed of at the Monkey Mountain Residue Area within the town of Port Hope between 1945 and 1948. Large quantities of waste from this site were removed and disposed of at Port Granby in 1959 and 1966. As a result of these previous Ra and U refining operations, Port Hope Harbour is contaminated by ^{238}U and ^{232}Th decay chain radionuclides.

Suspended U concentrations in air are measured at three sites in the area around the Cameco plant. A high-volume air sampler is operated for 24 hours, after which time the filter is analysed for U.

Data are available for ^{226}Ra and U concentrations at various locations in and near Port Hope Harbour in 1982 and at the cooling water intake from 1985 to 1998. Water quality is measured at the exit point to Lake Ontario near the plant's south cooling water intake on a daily basis. Cameco publishes its water quality monitoring data in Quarterly Environmental Compliance Reports. Uranium concentrations in water at the cooling water intake have decreased from the 1980s. However, sediment in the turning basin and west slip of Port Hope Harbour is still contaminated with radionuclides and other pollutants. The turning basin area is the most contaminated with respect to radionuclide concentrations, with concentrations in sediment decreasing with increasing distance from the effluent source.

Bottom-feeding fish, including the brown bullhead and yellow perch (*Perca flavescens*), have consistently higher radionuclide concentrations in their flesh than other open-water species (e.g., lake trout). The bullhead and perch are both potentially long-term residents of the turning basin, as both species prefer warm-water habitats and are tolerant of low dissolved oxygen conditions (which are common in the turning basin). Due to their feeding habits and potentially long residence times in the basin, the bullhead and perch are expected to have the greatest exposure to sediment radionuclide contamination, and consumption of these species by upper trophic levels (e.g., loons, raptors) may allow transfer of sediment radionuclide contamination to the terrestrial and/or aquatic food chains.

In 1987, surface soil (0–5 cm) was sampled from 23 sites that were selected based on a gradient of inorganic contamination identified in an earlier (1986) survey. A more recent survey of U levels in Port Hope soil was undertaken by the Phytotoxicology Section of the Ontario Ministry of Environment and Energy in 1997. Soil U concentrations are highest near the Cameco facility and decrease with increasing distance, from a maximum of 90 $g \cdot kg^{-1}$ dw to <2 $g \cdot kg^{-1}$ dw at sites farther from town.

The Low-Level Radioactive Waste Management Office (1995) collected soil samples from 35 locations in Port Hope, including the approximately 300 ha predicted to be the area affected by stack emissions from operation of the Ra refinery from the 1930s to the 1950s. The maximum soil U concentration measured was 51.9 $g \cdot kg^{-1}$ dw soil.

Temporal trends in U concentrations in maple (*Acer* spp.) tree foliage for several sites at different directions from the Port Hope U facility from 1980 to 1987 show a substantial decrease in U concentration in foliage with time, and concentrations are approaching background values at some sites.

Uranium concentrations in aquatic media and realistic RQs for Port Hope Harbour are presented in Table 12. EEVs are based on samples collected mostly between 1994 and 1995. Risk quotients greater than 1 indicate the risk for potential U toxicity to both wildlife and aquatic biota, particularly benthic invertebrates. Exposure is primarily by way of the sediment pathway and from ingestion of contaminated fish and macrophytes by wildlife.

Recent maximum U concentrations in soil near the Port Hope U refinery are given in Table 13. EEVs are based on data from samples collected in 1997. The conservative RQs for soil invertebrates (earthworms) indicate that U is not toxic to these organisms. However, the maximum U concentrations may be harmful to plants.

3.3.2.3.3 Port Granby Waste Management Facility, Ontario

The Port Granby Waste Management Facility (PGWMF) is operated by Cameco and is located about 16 km west of Port Hope. The facility occupies about 18 ha and borders Lake Ontario. The PGWMF received low-level radioactive waste from Eldorado (the predecessor company to Cameco) between 1955 and 1988, but is no longer accepting waste material today. The waste (process residue, scrap equipment, industrial trash and soils) is covered by at least 1 m of soil, which is planted with vegetation. A wastewater collection and treatment system collects and treats contaminated groundwater and surface water before releasing the water to Lake Ontario.

Radiation doses at the perimeter fence are monitored continuously using thermoluminescent dosimeters located at two sites — the east gate and west gate. Water quality is monitored in Lake Ontario during periods of overflow from the water treatment reservoirs.

Uranium concentrations in water and realistic RQs are presented in Table 12. On the basis of limited data and the high U concentrations reported in water during two overflow events, RQs greater than 1 indicate the intermittent potential for U effects on both wildlife and aquatic biota within the PGWMF during storm overflow events at East Reservoir. These elevated U concentrations are not harmful to aquatic biota in Lake Ontario because of rapid dilution of concentrations. However, there is the potential for localized effects near the confluence with Lake Ontario.

3.3.2.3.4 Welcome Waste Management Facility, Ontario

The WWMF occupies 36 ha of land and is located approximately 1 km west of Port Hope, Ontario. It is an L-shaped property that is bounded by township roads on the east and west, agricultural land on the south, Highway 401 on the north and an autowrecker's yard on the northeast.

The WWMF received low-level radioactive contaminated waste (process residue and ore rejects) from Eldorado facilities (i.e., Port Hope refinery) from 1948 to 1955. In 1955, the Welcome site was closed due to the contamination of a neighbouring stream and pasture with As, and a new waste disposal site was established at Port Granby. The total volume of waste received at Welcome over the period of operation was estimated to be 12 000 m^3. The average concentrations of U and Th in the material are approximately 140 and 8 $g \cdot kg^{-1}$, respectively. Thorium-230 and ^{226}Ra concentrations were around 52 and 61 $Bq \cdot g^{-1}$, respectively. The facility no longer accepts waste. By the mid-1980s, all wastes had been buried under layers of soil to improve their isolation, and soils were vegetated. In the 1980s, a program was undertaken to clean up soils on property around the WWMF that had been contaminated by wind and water erosion during the early years of the facility's operation.

A water collection system was built in 1956 to collect surface water and groundwater contaminated with U, Ra and As. This water was pumped directly into Lake Ontario with no prior treatment. In 1979, a treatment process was installed to reduce Ra and As loading to Lake Ontario. This system was improved in 1986–87 by increasing the volume of the collection pond and through construction of an interceptor ditch along the north side of the property.

Uranium concentrations in water and realistic RQs are presented in Table 12. EEVs are based on samples collected between 1993 and 1996. On the basis of limited data and the relatively high U concentrations reported in water, RQs greater than 1 for *Ceriodaphnia* and *Daphnia* indicate the potential for U effects on pelagic organisms such as zooplankton. Note that the RQs less than 1 for wildlife do not necessarily mean that U is not potentially harmful to these animals, but reflects the lack of data on sediment U concentrations.

3.3.2.3.5 Chalk River Laboratories, Ontario

Atomic Energy of Canada Limited's CRL is located near Chalk River, Ontario, on the Ottawa River. Chalk River Laboratories is the site of the National Research Universal (NRU) Research Reactor, the Molybdenum-99 Production Facility, the WMFs, the Universal Cells, and the Fuel and Material Cells. Atomic Energy of Canada Limited maintains an environmental monitoring program, which monitors airborne emissions and liquid effluents and samples gamma exposure, air, precipitation, drinking water, and both aquatic and terrestrial environments. There are several sources of liquid effluent discharges to the Ottawa River. These include discharge from the process sewer that carries cooling water and sump discharges from the NRU reactor, the Heavy Water Upgrading Plant and various facilities and decontaminated water from the Treatment Centre; discharge from the sanitary sewer; discharge from the 04 storm sewer; the 03 stream, which carries surface runoff and subsurface drainage from CRL Site Controlled Area 1; the 05 stream, which carries surface runoff and subsurface drainage mainly from the west parts of CRL Site Controlled Area 2; and Perch Creek, which drains several WMF facilities via Perch Lake.

Unlike Port Granby and Welcome, the WMFs at CRL do not contain appreciable quantities of U. Because of this, U is not a potential toxicant in the CRL WMF, and no U data are available. Therefore, RQs for the potential toxicity of U to wildlife and aquatic biota were not calculated for CRL.

3.3.2.4 Summary and conclusions

Realistic RQs were calculated to assess the potential toxicity of U released from U mines, U refinery and fabrication facilities, and WMFs to representative wildlife species and to aquatic biota. Pathways considered in the assessment are exposure to contaminated water and sediment and ingestion of contaminated food, macrophytes and fish. Ingestion of contaminated sediment/soil by wildlife in foraging and grooming was also considered in estimating exposure to U. The widespread occurrence of RQs greater than 1 for both wildlife and aquatic biota at U mines and mills indicates that the potential for harm from exposure to U released from U mines and mills is widespread. Potential for localized effects on both wildlife and aquatic organisms has also been identified at two of the three WMFs examined.

3.4 Ionizing radiation

Radiation is measured as the absorbed radiation dose, which is the amount of energy absorbed by an organism, organ or tissue due to exposure to ionizing radiation divided by the mass of the organism, organ or tissue ($J \cdot kg^{-1}$) in grays (Gy), where 1 Gy = 1 $J \cdot kg^{-1}$. Ionizing radiation has an energy of at least 12.4 eV and has the ability to remove an orbital electron from an atom to form an ion-electron pair from a neutral ion. The nucleus of radioactive materials is not stable and transmutes or decays to a different nucleus with the release of energy in the form of alpha, beta or gamma radiation. These decay at a rate dependent on the half-life of the radionuclide.

Alpha radiation is produced mainly from atoms of high atomic mass. Decay of the nucleus is by emission of a fast-moving helium nucleus, an alpha particle. Alpha particles travel a short range (<15 cm) and do not penetrate the dead layer of skin (~70 µm thick). Alpha radiation is an internal hazard.

A beta particle is a free electron or positron emitted by radionuclides during radioactive decay. Beta particles can penetrate biological tissue to a depth of 1 or 2 cm. They may pose both an internal and an external hazard.

Gamma radiation is the emission of photons from the nucleus. Gamma radiation is much more penetrating than alpha and beta radiation and has no precise range. The intensity of gamma radiation attenuates exponentially with distance in dense media. Gamma radiation is both an external and internal hazard.

Radiations differ in their relative biological effectiveness (RBE) per unit of absorbed dose. A weighting factor is used to account for this difference in RBE of different types of radiation at low doses for biological effects such as cancer induction and genetic defects. A weighting factor of 1 is used for gamma and beta emissions, except for beta emissions from ^{3}H, which has a weighting factor of 3, and a weighting factor of 40 is used for alpha emissions (Section 3.4.1.8).

To assess the effect of radionuclide releases from nuclear facilities on non-human biota, it is the chronic exposure to low doses from routine operations that is of interest. All dose estimates from experimental data and those made in the present report represent incremental doses above natural background. The radiation dose from ^{40}K, which represents the bulk of the natural background dose, was not included in the dose calculations. Elevated natural background dose due to U decay chain products near U mines and mills was considered, in that the radiation dose was calculated from pre-operational data for several mine sites (Section 3.4.2.1).

The naturally occurring alpha-emitting radionuclides appear to be the most significant sources of background radiation exposure for the majority of organisms. In the terrestrial environment, the source is ^{222}Rn and its short-lived decay products, whereas in aquatic environments, ^{210}Po is the major contributor (UNSCEAR, 1996).

Total radiation doses from background radiation have been found to be in the range of 0.6–7 $mGy \cdot a^{-1}$ for leaves and needles of terrestrial plants, 1–5 $mGy \cdot a^{-1}$ for terrestrial mammals and 0.7–1.7 $mGy \cdot a^{-1}$ for freshwater organisms (Macdonald *et al*., 1996; UNSCEAR, 1996). However, some species of wildlife may receive much higher doses of background radiation because of their habitat. For example, the accumulation of ^{210}Pb and ^{210}Po in the liver and kidney of some herds of Canadian caribou (*Rangifer tarandus*) (Thomas *et al*., 1992) may result in doses 10–100 times higher. Similarly, burrowing animals living in radon-rich soils may receive short-term exposures equivalent to over 100 $mGy \cdot a^{-1}$ (Macdonald and Laverock, 1998).

3.4.1 Effects characterization

The effects of exposure to radiation (Table 14) have been reviewed by several international agencies for the purpose of identifying a radiation dose below which effects on populations of organisms would not be likely to occur (IAEA, 1976, 1992; NCRP, 1991;UNSCEAR, 1996). These reports, using the experts review approach, have suggested that doses of approximately 10 $mGy \cdot d^{-1}$ (3.7 $Gy \cdot a^{-1}$) and 1 $mGy \cdot d^{-1}$ (0.4 $Gy \cdot a^{-1}$) for aquatic and terrestrial species, respectively, would not put populations at risk. These radiological protection standards are based on the assumption that populations of organisms possess compensatory capabilities such that impacts on a few individuals should not affect the integrity of a population or community (Barnthouse, 1997).

The ecological risk assessment approach for Priority Substances used in this assessment is somewhat different from that used to derive these radiation protection standards. The approach used in this assessment requires identifying CTVs from which ENEVs are derived using appropriate application factors (Environment Canada, 1997).

The preferred CTVs are estimates of low toxic effects, such as the LC_{25} or EC_{25} for the most sensitive species (Environment Canada, 1997). If the most sensitive species is protected, it is given that other less sensitive taxa will also be protected. CTVs for the various taxonomic groups are based on the most sensitive response applicable to the survival of the species and therefore may include any one of these endpoints and are based on chronic exposures. A relatively large amount of information is available on the effects of radiation on the environment in comparison with many other contaminants, with data being available for most major taxonomic groups. Therefore, application factors used may be less than or equal to 10.

The following sections briefly review the relevant radiation effects data and identify CTVs for both aquatic and terrestrial organisms. More detailed information on the effects of radiation exposure can be found in ECOMatters Inc. (1999a), Harrison (1997), Macdonald (1999), Anderson and Harrison (1986) and the reviews cited above. Exposure to radiation is estimated by summing all external (e.g., from soil/sediment, air, water, etc.) and internal (e.g., ingested, internally deposited, inhaled, etc.) sources of radiation and relating the dose with dose–response data (i.e. ENEVs) (see Section 3.4.1.6).

At high doses, effects are expressed by the death of cells, which may ultimately result in loss of tissue and organ function and, if the dose is high enough, the death of the organism. Major systems affected by acute exposures are the hematopoietic, gastrointestinal and immune systems. Lower doses and dose rates affect reproduction and can cause chromosomal aberrations and mutations. Genetic effects, such as point mutations, single strand breaks (SSBs), double strand breaks (DSBs) and sister chromatid exchanges, occur at lower doses in both somatic and germinal cells and may be expressed at the population level.

Genetic effects are the major consequence from radiation exposure at low and moderate dose rates. Radiation causes many types of damage to genetic material. Small errors in DNA repair and changes in the gene expression regulatory mechanisms can lead to cancer. Cancer is the primary concern in human radiation protection strategies. Non-human species also develop cancers in response to radiation (Gilbert *et al*., 1996). However, environmental radiation protection strategies focus on the effects on reproduction, because wildlife populations typically overcome environmental stresses by increasing reproductive rates, when possible. This argument may not apply to long-lived, slowly reproducing species. Reproduction (processes from gametogenesis to embryonic development) is considered to be the most likely limiting endpoint in terms of survival of the population (Harrison, 1997). Therefore, the endpoints most relevant to ecological risk assessment are those that measure changes in the ability to reproduce, i.e., the factors that affect fertility and sterility, such as reduction in number of gametes produced, gamete death, and increased abnormalities and mortality of early life stages (Harrison, 1997). Genetic damage *per se* is not considered a limiting endpoint because of the difficulty in interpreting the significance of the effect at the population level (i.e., population fitness and survival), and it is assumed that genetic effects are integrated in reproductive effects. The lack of consideration of genetic damage *per se* in this assessment is consistent with the approach used for other potentially genotoxic Priority Substances.

The extent and type of damage from radiation exposure depend on the type of radiation and the amount of energy absorbed at the site of impact. Gamma and X-ray radiation penetrate through biological tissue, deposit less energy (i.e., low linear energy transfer [LET]) and tend to produce SSBs in chromosomes. Alpha particles penetrate only about 70 µm into tissues, deposit much more energy at the site of impact (i.e., high LET) and tend to cause DSBs and more rapid cell death.

Cells that are not dividing are more resistant to ionizing radiation than cells that are replicating. Rapidly dividing cells are most sensitive, suggesting that chromosomal structure is most vulnerable during division. Presumably, this is because this is a time when priorities for division take precedence over activation of repair mechanisms. Thus, cell absorption of ionizing radiation leads to abnormal mitosis, growth and metabolism. Mutations are proportional to the amount of radiation absorbed, depending upon the amount of DNA in the cell, the functioning state of the cell and the effectiveness of the repair mechanisms. Generally, a higher dose rate will cause a greater impact than a lower dose rate, giving the same total dose. This is assumed to be because repair mechanisms are better able to keep pace with damage caused by a lower dose rate.

3.4.1.1 Mammals

There are numerous studies concerning the effects of chronic radiation exposure on mammals. The data are extremely variable. The IAEA (1992) concludes that a dose rate of 10 $mGy \cdot d^{-1}$ (3.7 $Gy \cdot a^{-1}$) is a threshold at which reproductive capacity is affected; acute doses of 0.1 Gy are very unlikely to produce persistent, measurable deleterious changes in populations or communities of terrestrial plants or animals; and irradiation at a rate of 1 $mGy \cdot d^{-1}$ (0.4 $Gy \cdot a^{-1}$) is unlikely to cause observable changes in terrestrial

animal populations. UNSCEAR (1996) concluded that dose rates below 10 mGy·d^{-1} (3.7 Gy·a^{-1}) to the most exposed members of the population would not seriously increase the death rate of mammal populations, and reproductive effects are unlikely at 10% of the lethal dose rate, or 1 mGy·d^{-1} (0.4 Gy·a^{-1}). Furthermore, a dose rate of 0.9 Gy·a^{-1} to the most highly exposed individuals is unlikely to significantly affect the fecundity of mammalian populations. UNSCEAR (1996) also cites a study in which the developing oocytes of the squirrel monkey (*Saimiri sciureus*) had an LD_{50} of 42 µGy·d^{-1} (15 mGy·a^{-1}), giving total doses in the range of 40–200 mGy over the study. As a general rule, chronic effects begin at 10% of the LD_{50} value, and effects on oocytes occur at 1% of the LD_{50}. Rose (1992) concluded that the lowest exposure rate producing a spectrum of effects is around 1 Gy·a^{-1}, although effects have been observed at lower doses. For example, a dose of 0.07 Gy·a^{-1} increased the mortality of laboratory mice offspring using tritiated water (Dobson, 1982), and a dose of 0.35 Gy·a^{-1} reduced testicular mass and epididymal sperm counts after 30 weeks' exposure to alpha radiation from plutonium (Searle *et al.*, 1976). In contrast, no effects were observed on the fertility of rats exposed to 36 Gy·a^{-1} (Brown *et al*., 1964).

Most results for mammals indicate that natality is a more radiosensitive parameter than mortality. The total accumulated dose at which a given response occurs increases as the dose rate decreases and as the exposure is protracted. The most sensitive endpoint for mammals appears to be the killing of 50% of immature oocytes, which in itself does not result in sterility, by 1 mGy·d^{-1} (0.4 Gy·a^{-1}) during the last trimester of fetal development in monkeys (Dobson, 1982). Slight changes in bull semen occur at an acute dose of 0.5 Gy, but recovery is complete in about 30 weeks post-irradiation. Mice are among the most sensitive species to reproductive effects of radiation; reproduction is impaired by an acute dose as low as 0.2 Gy for females, while males are less sensitive (3.2 Gy) (IAEA, 1992). A dose of 3.1 mGy·d^{-1} (1.1 Gy·a^{-1}) to neonatal mice from ^{3}H produced a 50% reduction in number of immature oocytes. Note that more immature oocytes are produced than can be utilized for reproduction, so this may not affect reproduction. Chronic exposure to gamma radiation at a rate of 13–26 mGy·d^{-1} (4.7–9.5 Gy·a^{-1}) over 10 generations did not produce changes in litter size of mice or sex ratios of progeny. Exposure of male mice to high doses (9 Gy for each of 8 generations, 3 Gy for 15 generations, or 2 Gy for 35 generations) resulted in no change in health or fitness of the offspring. Survival of mammals in an irradiated hardwood forest was not affected at a dose rate of 20 mGy·d^{-1} (7.3 Gy·a^{-1}) (Buech, 1977).

The lowest acute dose in the literature that caused sublethal effects is 10 mGy to pregnant rats, which impaired the reflexes of their offspring (Rose, 1992). However, increases in fertility have also been observed at low dose rates, and normal breeding mice are reported at 16 Gy·a^{-1} (Golder Associates Ltd., 1996). No effect was observed on the fertility of 10 generations of rats exposed to 7 Gy·a^{-1} or for six successive litters produced by female rats exposed to 36 Gy·a^{-1} (Brown *et al*., 1964). Wild rodents living on the dry bed of White Oak Lake exposed to lifetime doses of 2–3 Gy showed no effects that could be ascribed to radiation (Dunaway and Kaye, 1963). However, an increase in micronuclei in the bank vole (*Clethrionomys glareolus*) was observed at dose rates up to 14.4 mGy·a^{-1} (Cristaldi *et al*., 1991). Similar chromosomal effects were observed in caribou in Norway after

Chernobyl at doses of 50–60 mGy (Røed *et al.*, 1991). Evidence is also accumulating that damage occurs to genetic material that is not expressed in the irradiated generation, but may be expressed several generations later (Mothersill and Seymour, 1997).

From the data presented above, the LD_{50} of 1 $mGy \cdot d^{-1}$ (0.4 $Gy \cdot a^{-1}$) for the squirrel monkey (Dobson, 1982) was chosen as the CTV. An application factor of 1 was used to set a realistic ENEV of 0.4 $Gy \cdot a^{-1}$. This ENEV, based on a small mammal study, is not considered overly conservative, since effects data are relatively abundant, and the available data suggest that small mammals (which are the focus of this assessment) are less sensitive to the effect of radiation than large mammals (e.g., moose).

3.4.1.2 Amphibians and reptiles

Data on the effects of radiation on survival are generally available only from studies with acute exposures where the post-irradiation observation period is often 30 days (30-d LD_{50}). Extending the observation period usually lowers the dose causing 50% mortality. In the case of poikilothermic (cold-blooded) animals, temperature can control the time of expression of radiation effects. Therefore, for fish, amphibians and reptiles, which are poikilotherms, a 60- or 90-day study period is more appropriate than the 30-day period normally employed for mammals.

Although both reptiles and amphibians appear to be less sensitive to radiation than mammals (Ewing *et al*., 1996; UNSCEAR, 1996), work by Sparrow *et al*. (1970) indicates that the LD_{50} for the mud puppy (*Necturus maculosus*) is less than 1 Gy, putting it in the same range of sensitivity as mammals. Many LD_{50}s reported in the literature are not directly comparable to those for mammals, because death in amphibians occurs well after the 30-day period (i.e., 30-d LD_{50}) generally used for comparison (Sparrow *et al*., 1970). This is clearly shown for the mud puppy in experiments in which the 30-d LD_{50} of 35.5 Gy drops to 0.8 Gy 200 days after exposure (i.e., 200-d LD_{50}). It is hypothesized that *Necturus* is more sensitive than other amphibians because of a very large interchromosomal volume (Conger and Clinton, 1973) and lack of a suitable system to repair the radiation damage (Sparrow *et al*., 1970). LD_{50}s for adult anurans (frogs and toads) range from about 6 to 20 Gy (Ewing *et al*., 1996). Panter (1986) showed that the most sensitive stage for acute exposure for the frog (*Limnodynastes tasmaniensis*) is the fertilized egg, with a 40-d LD_{50} of 0.6 Gy. Urodeles (e.g., newts, mud puppy) are also sensitive with LD_{50} values between 1 and 5 Gy, assuming up to 300 days post-exposure for the time of lethality (Sparrow *et al*., 1970). Juvenile life stages have lower LD_{50}s, ranging as low as 0.9 Gy for Fowler's toad (*Bufo woodhousei fowleri*). The LD_{50} for acute radiation changes markedly through the developmental stages of a frog, increasing from <1 Gy in the early stages of development to over 25 Gy in the adult (Panter *et al.*, 1987). LD_{50}s for reptiles are in the same range as those for adult amphibians (>8 Gy).

The lowest acute LD_{50} reported is 0.6 Gy to the fertilized eggs of the frog *L. tasmaniensis*. The next lowest acute LD_{50} reported is a 200-d LD_{50} of 0.8 Gy to the mud puppy (Sparrow *et al*., 1970). Because the mud puppy is native to the Canadian Shield, whereas *L. tasmaniensis* is an exotic species,

the toxicity data for the mud puppy (Sparrow *et al.*, 1990) were used in setting an ENEV for amphibians. Using an acute:chronic toxicity ratio of 10, a chronic toxicity value of 0.08 Gy is obtained. This value was taken as the CTV. A realistic ENEV is obtained using an application factor of 1 to give an ENEV of 0.08 $Gy \cdot a^{-1}$.

3.4.1.3 Fish

Several reviews on the effects of radiation on aquatic organisms have been published (Polikarpov, 1966; Templeton *et al.*, 1971; Chipman, 1972; Ophel, 1976; Blaylock and Trabalka, 1978; Egami and Ijiri, 1979; Woodhead, 1984). Anderson and Harrison (1986) synthesized the data on effect levels for a number of endpoints from studies having appropriate dosimetry. Their review indicated that a dose rate of 5–100 $mGy \cdot d^{-1}$ (1.8–37 $Gy \cdot a^{-1}$) would encompass the level at which a variety of low-level effects on reproduction, development and genetic integrity are detectable in sensitive tissues and organisms. The IAEA (1988) concluded that reduced reproductive success would likely occur at dose rates in the range of 24–240 $mGy \cdot d^{-1}$ (8.8–88 $Gy \cdot a^{-1}$).

Fish are the most radiosensitive of the non-mammalian aquatic organisms, with reproductive capacity being the most sensitive endpoint. Other aquatic life is relatively insensitive to chronic irradiation. For example, for *Daphnia pulex*, the threshold effect level for mortality is 960 $mGy \cdot d^{-1}$ (350 $Gy \cdot a^{-1}$) under the additional stress of food limitation (see Section 3.4.1.4.2). It should be noted that earlier studies reporting radiation effects in fish at very low doses are considered unreliable because the findings could not be reproduced, and many of the earlier studies were poorly designed, particularly with respect to incubation techniques. Larval abnormalities were attributed to radiation but were most likely due to poor rearing techniques (Rose, 1992).

The LD_{50} for adult fish ranges from 3.8 to 30 Gy, whereas the LD_{50} of fish embryos ranges from 0.16 to 25 Gy (Table 15). The lowest LD_{50} for fish reported was 0.16 Gy in coho salmon (*Oncorhynchus kisutch*) exposed at the one-cell stage and observed for 150 days (Bonham and Welander, 1963). The next lowest LD_{50} was 0.58 Gy for the one-cell stage of rainbow trout (*Oncorhynchus mykiss*) (Welander, 1954), then 0.9 Gy from exposure to X-rays for plaice (*Pleuronectes platessa*) larvae irradiated at the blastula stage (Ward *et al.*, 1971). Chinook salmon (*Oncorhynchus tshawytscha*) embryos irradiated at 0.5 $mGy \cdot d^{-1}$ (0.2 $Gy \cdot a^{-1}$) for 20 days showed no increase in mortality until the time they were released as smolts (Bonham and Donaldson, 1966). The lowest acute exposure causing effects on reproductive tissue of fish appears to be 0.25 Gy (Anderson and Harrison, 1986).

Dose rates at which detrimental effects on fertility are first observed in fish are similar to those observed in mammals (i.e., 5–100 $mGy \cdot d^{-1}$) (Anderson and Harrison, 1986; Woodhead and Pond, 1987). Few studies have been conducted to determine radiation doses that would cause mortality in fish as a result of chronic exposures. Significant reductions in fecundity have been observed at chronic doses ranging from <14.4 to 312 $mGy \cdot d^{-1}$. Chinook salmon embryos exposed for 16–20 days and for

periods up to 80 days to an external ^{60}Co source at a dose rate of 5 mGy·d^{-1} (total dose of 330–400 mGy) showed a significant increase in opercular defects, but did not show any significant differences in mortality relative to controls — i.e., abnormalities in young fish increased, but the number of adult salmon returning was not affected (Donaldson and Bonham, 1964). Irradiation of salmon eggs at 5 mGy·d^{-1} (1.8 Gy·a^{-1}), with a total dose of 3.6 Gy, produced no damage to salmon populations sufficient to reduce the reproductive capacity over a period of several generations. It is noteworthy that fish receiving a low radiation dose returned in greater numbers and produced a greater total number of viable eggs than did non-irradiated control fish. At 100 mGy·d^{-1}, radiation damage was evident, and the growth rate of these fish was significantly less than that of controls (Marko, 1981).

A significant reduction in growth rate was observed when rainbow trout embryos were acutely exposed to 0.38 Gy (Welander, 1954). In the same study, an increased frequency of abnormalities was observed in embryos irradiated at 2 Gy. Major malformations were observed when developing eggs of the mummichog (*Fundulus heteroclitus*) were exposed to 3–4 Gy (Rugh and Clugston, 1955). An acute dose of 5 Gy caused a 50% reduction in hatching of carp (*Cyprinus carpio*) eggs (Blaylock and Griffith, 1971). In mosquitofish (*Gambusia affinis*) living in White Oak Lake (Oak Ridge National Laboratory), a pond contaminated with radionuclides, exposure to 0.6 mGy·d^{-1} (0.2 Gy·a^{-1}) over a lifetime produced a significant increase in embryo mortality (Trabalka and Allen, 1977). In contrast, exposure of young guppies (*Poecilia reticulata*) to a total dose of 3.4–47 Gy from ^{3}H did not result in a significant increase in mortality (Erickson, 1973).

Aquatic populations may be expected to experience increased mortality at sustained dose rates of 0.24 Gy·d^{-1} (88 Gy·a^{-1}) and reduced reproductive success at 0.024–0.24 Gy·d^{-1} (8.8–88 Gy·a^{-1}). At lower doses, the minor effects that would occur in individuals would likely be accommodated within the reproductive capacities of populations or eliminated through natural selection (Whicker and Schultz, 1982; McKee *et al*., 1988). However, there are few data on less fecund, slower-growing fish species typical of northern waters, such as lake trout and walleye (*Stizostedion vitreum*) (Golder Associates Ltd., 1996).

For fish, the lowest acute LD_{50} is 0.16 Gy in coho salmon exposed in the one-cell stage, whereas the lowest chronic toxicity reported was 0.6 mGy·d^{-1} for the mosquitofish. The value of 0.6 mGy·d^{-1} was chosen as the CTV. This value is actually a NOEC — i.e., although there was a significant increase in mortality from 3% for controls to 5–6% for exposed embryos, this increase is considered not to be biologically significant. An application factor of 1 was used for the ENEV, which results in a value of 0.6 mGy·d^{-1} (0.2 Gy·a^{-1}).

3.4.1.4 Invertebrates

3.4.1.4.1 Terrestrial

Adult insects are relatively hardy when it comes to radiation exposure, because very little cell division and differentiation occur in adults. Juvenile stages of insects are much more sensitive to radiation because of higher rates of cell turnover and differentiation. About 0.1 Gy kills the eggs of the braconial wasp (*Bracon hebetor*) (O'Brien and Wolfe, 1964), and <1.3 Gy kills housefly (*Musca domestica*) eggs (Cole *et al*., 1959). Experimental addition of radionuclides to soil pots reduced soil invertebrate numbers only at high doses (0.5–1.9 $Gy \cdot d^{-1}$), although earthworm (Lumbricidae) populations were reduced at 24 $mGy \cdot d^{-1}$ (Krivolutsky, 1987). Invertebrates appear to be more affected by indirect effects of chronic exposure such as loss of litter than to irradiation itself.

The lowest effect value reported for terrestrial invertebrates was 24 $mGy \cdot d^{-1}$ (8.8 $Gy \cdot a^{-1}$) for effects on earthworms. This value was adopted for the CTV. An application factor of 10 was used to account for possible effects on reproduction to give an ENEV of 2.4 $mGy \cdot d^{-1}$ (0.88 $Gy \cdot a^{-1}$).

3.4.1.4.2 Aquatic

Limited data are available on the effects of acute radiation exposure on freshwater invertebrates (Anderson and Harrison, 1986). In invertebrates, significant reductions in fecundity have been observed at chronic doses ranging from 1.7 to 13 200 $mGy \cdot d^{-1}$ (0.6 to 4800 $Gy \cdot a^{-1}$). Embryos of the goose barnacle (*Pollicipes polymerus*) were most sensitive to radiation, showing a reduction in moulting when exposed to tritiated water at a dose rate of 1.7 $mGy \cdot d^{-1}$ (0.6 $Gy \cdot a^{-1}$) (Abbott and Mix, 1979). For aquatic ecosystems, benthic organisms are likely to be the most highly exposed organisms due to the partitioning of radionuclides to sediment. Chronic irradiation at about 2 $Gy \cdot a^{-1}$ to *Chironomus tentans* larvae in White Oak Lake increased the frequency of chromosomal aberrations, but had no apparent additional effects on the population (Blaylock, 1965). Reduction of the dose rate to about 0.1 $Gy \cdot a^{-1}$ due to radiological decay resulted in the frequency of chromosomal aberrations decreasing to that found in reference populations. Also in White Oak Lake, the snail, *Physa heterostropha*, experienced a reduction in egg capsule production at a radiation dose of about 6 $mGy \cdot d^{-1}$ (2.2 $Gy \cdot a^{-1}$), but there was no difference in production from the non-irradiated population because the number of eggs per capsule increased in the irradiated population (Cooley and Nelson, 1970; Cooley and Miller, 1971; Cooley, 1973). In the laboratory, a dose rate of 240 $mGy \cdot d^{-1}$ (88 $Gy \cdot a^{-1}$) from ^{60}Co had no significant effect on reproduction, mortality or size of the snail (Cooley and Miller, 1971). However, a dose rate of 2.4 $Gy \cdot d^{-1}$ (880 $Gy \cdot a^{-1}$) had significant effects. The pond snail (*Physa acuta*) exposed as a four-celled embryo had an acute LD_{50} of 10.8 Gy (Ravera, 1968).

Sublethal effects in invertebrates are generally not observed until doses are very high. For example, the growth rate of *Daphnia pulex* increased at 3–6.5 $Gy \cdot a^{-1}$ (Marshall, 1962). The lowest

dose causing developmental changes in invertebrates is 10 Gy for the calanoid copepod *Diaptomus clavipes*, which experienced a significant decrease in percent hatch (Gehrs *et al*., 1975). The data on marine invertebrates suggest that acute LD_{50}s range from 2.1 Gy (*Palaemonetes pugio*) to 560 Gy (*Callinectes sapidus*, adults) (Anderson and Harrison, 1986).

Of the invertebrates studied, the goose barnacle was the most sensitive to radiation, showing a decrease in reproduction at 1.7 $mGy \cdot d^{-1}$ (0.6 $Gy \cdot a^{-1}$). This value was chosen as the CTV. An application factor of 1 was used with the CTV to give an ENEV of 1.7 $mGy \cdot d^{-1}$ or 0.6 $Gy \cdot a^{-1}$.

3.4.1.5 Plants

3.4.1.5.1 Terrestrial

Radiation biology in plants has focused on mutations in order to produce more productive plants. Young plants are much more susceptible than mature plants (Mericle *et al*., 1955). The most noticeable location for radiation effects is the growing tip of the main root or stem. In general, plants that have many small chromosomes are more resistant to ionizing radiation than plants with a few large chromosomes. Woody species tend to be about twice as sensitive as herbaceous species, for a given interchromosomal volume (Whicker and Schultz, 1982). Low-stature plants and dormant seeds are more resistant to radiation effects (Whicker and Schultz, 1982).

The LD_{50} values for 60 woody plants range from 4.1 Gy for sugar pine (*Pinus lambertiana*) to 77 Gy for bitternut hickory (*Carya cordiformis*) and mockernut hickory (*C. tomentosa*) (ECOMatters Inc., 1999a). Angiosperms (conifers) are more sensitive, by almost an order of magnitude, than deciduous trees and are among the most sensitive of all plants.

There are relatively few studies involving chronic exposure of plants to radiation (ECOMatters Inc., 1999a). This is because of the logistic difficulties in having plants grow for an extended time in elevated radiation fields. Chronic studies are primarily from field irradiators placed outdoors and from areas contaminated by nuclear releases, such as around Chernobyl and at older, inadequate waste disposal sites. Long-term irradiation experiments with plants showed that the main endpoint is the loss of viable plants or mortality. The order of increasing radioresistance is as follows: coniferous trees, deciduous trees, shrubs, herbaceous species, lichen and fungi. A few data indicate that the production of viable seed is at least as sensitive an indicator of radiation damage as mortality (Woodhead, 1997).

The most sensitive study for which dose rate was directly measured involved a doubling of the needle length of Scotch pine (*Pinus sylvestris*) compared with the controls at a dose rate of 0.6 $mGy \cdot d^{-1}$ (0.2 $Gy \cdot a^{-1}$) in the Field Irradiation Gamma (FIG) field irradiator study in Manitoba (Sheppard *et al*., 1982). The next most sensitive study was a small (13%) detrimental effect on

growth observed in Scotch pine seedlings grown in small pots above a WMF at a dose rate of 2.4 $mGy \cdot d^{-1}$ (0.88 $Gy \cdot a^{-1}$) (Chandorkar and Dengler, 1987). The study of Amiro (1994) and Amiro and Sheppard (1994) reported a No-Observed-Effect Level (NOEL) of 2.4 $mGy \cdot d^{-1}$ (0.88 $Gy \cdot a^{-1}$) from the FIG field irradiator study. All of these studies reported effects at dose rates that are low compared with the literature, and all dealt with native Canadian plant species. These studies and those of Gunckel and Sparrow (1961), Sparrow *et al*. (1965) and Whicker and Fraley (1974) all reported detrimental effects at dose rates between 2.4 and 19 $mGy \cdot d^{-1}$ (0.88 and 7 $Gy \cdot a^{-1}$).

The lowest observed effect of acute irradiation was a radiation effect on an enzyme related to auxin, a growth regulator hormone in plants. An effect on activity was observed at 0.1 Gy (Gunckel and Sparrow, 1961). An increase in root growth of seeds was observed at 0.2 Gy, but whether this was a radiation response was ambiguous. A reduction in seed yield of 50% was observed at an acute dose of 0.51 Gy to a flowering plant. Other studies reported effects at dose levels of 1.2 Gy and above (Table 6.17 of Bird *et al*., 2000).

The NOEL of 2.4 $mGy \cdot d^{-1}$ reported by Amiro (1994) and Amiro and Sheppard (1994) was chosen as the chronic CTV. This value was measured in the field with native Canadian plants over a number of years. Given the very large number of species tested in the literature, it is unlikely that there will be many, if any, more sensitive species found in the future. With this level of confidence, rather more than is found in most ecotoxicology investigations, it is appropriate to use a low application factor. As a result, an application factor of 1 was used to derive an ENEVof 2.4 $mGy \cdot d^{-1}$ (0.88 $Gy \cdot a^{-1}$).

3.4.1.5.2 Aquatic

The lowest dose causing sublethal effects on aquatic plants (macrophytes and algae) is 0.07–0.12 $mGy \cdot d^{-1}$ (0.03–0.04 $Gy \cdot a^{-1}$), which caused a loss of synchrony in growth of *Chlorella pyrenoidosa* cultures (Chandorkar and Szachrajuk, 1978). Because of the scarcity of data for radiation effects on aquatic plants, the ENEV for conifers (terrestrial plants) of 0.88 $Gy \cdot a^{-1}$ was adopted as a realistic ENEV for both algae and macrophytes. Conifers are more sensitive to radiation than lichen (IAEA, 1992). Lichen are composed of a fungus in symbiotic union with an alga. Therefore, the use of conifer data for the ENEV is probably conservative.

The ENEVs for both terrestrial and aquatic organisms used in this assessment are summarized in Table 16.

3.4.1.6 Calculation of risk quotients for potential radiation effects

To assess the potential harmful effects of contaminant concentrations at the various nuclear facilities, two sets of calculations were performed to estimate the RQ, one using very conservative values and the other using more realistic values. In the first set of calculations, hyperconservative EEVs were used to screen for potential toxic effects. If the resultant RQs were less than 1, the releases of radionuclides are

considered not toxic under CEPA 1999 for radiation effects, and no further calculations were performed. If RQs were greater than 1 for any given taxonomic group of organisms, then a second set of calculations was performed for that taxonomic group (RQ >1) using realistic values for exposure (EEVs). Conservative screening estimates used the most recent maximum radionuclide concentrations reported in the environment or, in some cases, in the effluents. Where concentrations in biota were not measured, they were estimated by multiplying the maximum concentration in a given medium by the appropriate maximum CRs in Bird and Schwartz (1996). If CRs for a given radionuclide were not available in Bird and Schwartz (1996), then the CR values presented in Davis *et al*. (1993) were used (e.g., for ^{210}Po). In most cases, data used were from 1996–1998. However, in certain instances, older data were used. For example, radionuclide concentrations reported for Port Hope Harbour in 1984 were used in the present assessment because these values were the most recent information available. When no data were available for a daughter radionuclide, the daughter was assumed to be in secular equilibrium with its parent radionuclide. For example, ^{210}Bi was assumed to be in secular equilibrium with ^{210}Pb.

In most cases, data were available for radionuclide concentrations in fish and macrophytes. For fish collected at U mines, it was a common practice to present radionuclide concentrations in both flesh and bone. In this case, the radiation dose to the bone was added to the radiation dose to the flesh. This approach leads to a conservative dose estimate, because bone tends to have higher concentrations than flesh. The total radiation dose to fish was the sum of the internal dose and external dose from water and from sediment. The internal radiation dose was calculated based on the measured radionuclide concentrations in fish or those estimated using CRs between fish and water. In the calculation of risk quotients for potential radiation effects, the EEV is expressed as the total radiation dose (internal dose plus external dose). In this assessment, internal radiation doses were estimated using tissue concentrations of radionuclides and internal dose conversion factors (DCFs) of Amiro (1997). The external dose was estimated using radionuclide concentrations in the external medium (e.g., sediment, water) and external dose factors of Amiro (1997).

The radiation dose to macrophytes was calculated in a manner similar to that for fish, i.e., the total dose was the sum of the internal dose and the external dose from water and from sediment. The internal dose was based either on measured radionuclide concentrations in macrophytes or on values estimated using a CR between water and aquatic macrophytes. No measurements were available for radionuclide concentrations in algae (phytoplankton). For this reason, CRs were used in all cases to estimate the radionuclide concentrations in algae. The total dose to algae was the sum of the internal radiation dose and the external dose from water. The external dose from sediment was not included in the total radiation dose to algae.

For the radiation dose to benthic invertebrates, it was assumed that ionization was uniform throughout the sediment. Chironomids and tubificid worms often comprise a large portion of the benthic fauna. These organisms are small and may be viewed as tiny cylinders surrounded by sediment both externally and internally (i.e., the gut is filled with sediment). They are approximately 90% water (body

fluids) and 10% tissues. Likewise, the surficial sediments are mostly water. The invertebrates receive an internal dose from radionuclides deposited internally (in their tissues and cytoplasmic fluids) as well as from those radionuclides sorbed to both the external cuticle and internal cuticle lining their gut. This approach departs from the standard approach for assessing the toxicity of metals to benthic invertebrates, which accounts for the bioavailability of the contaminants. This is because external radionuclides possibly associated with the cuticle may cause considerable local radiation damage, whereas non-radioactive external contaminants are not usually toxic. This approach also departs from the standard approach of estimating the radionuclide concentration in benthic invertebrates based on water concentrations and CRs. This standard approach has generally been used in situations where aqueous radionuclide concentrations are available through either measurements or model predictions, but sediment concentrations are not. However, this approach may lead to greater uncertainty in radiation exposure to benthic invertebrates that live in the sediment, as opposed to on the sediment, because of the great variability in CRs. CRs tend to be relatively site specific and tend to decrease as aqueous concentrations increase. The choice of a CR is particularly difficult when CRs are applied to aqueous concentrations that are below the detection limit. Estimates of radiation exposure based on sediment concentration are more realistic than estimates based on aqueous concentrations, and this is the method of choice for this assessment.

Maximum CRs and GM CRs between water and fish, algae and macrophytes are presented in Table 17.

The internal radiation doses to terrestrial organisms — i.e., small mammals, lichen, blueberry (*Vaccinium* spp.) bushes and Labrador tea (*Ledum groenlandicum*) — were based on measured radionuclide concentrations in these organisms. Radionuclide concentrations in soil and litter invertebrates were derived from soil radionuclide concentrations following the same approach used for benthic invertebrates living in sediment.

3.4.1.7 Dose conversion factors (DCFs) and assumptions

Internal doses from individual radionuclides were calculated by multiplying the radionuclide concentration in the organism (estimated or measured) by the internal DCF given in Amiro (1997), whereas external doses were calculated by multiplying the water or sediment radionuclide concentration by the appropriate external water or soil/sediment DCF. For those radionuclides (^{51}Cr, ^{54}Mn, ^{60}Co, ^{95}Zr, ^{95}Nb, ^{106}Ru, ^{124}Sb, ^{134}Cs, ^{144}Ce, ^{144}Pr and ^{228}Ac) for which no DCFs were given by Amiro (1997), DCFs were calculated following the method outlined in Amiro (1997). The internal DCF values of Amiro (1997) do not account for the RBE of different types of radiation emissions (i.e., do not use a radiation weighting factor). Therefore, the DCFs were modified to account for the RBE of alpha and beta radiation in the case of ^{3}H. Weighting factors of 40 and 3 were used for alpha radiation and for beta radiation from ^{3}H, respectively. The DCFs used in this assessment are presented in the supporting document (Bird *et al*., 2000).

The internal DCF values in Amiro (1997) represent organisms of all sizes and conservatively assume that all energies emitted by radionuclides from within the organism are also absorbed by the organism. This is reasonable for alpha particles and electrons that do not travel far, but will overestimate the dose from photons. This overestimation is small for large organisms, because much of the energy will be absorbed by the large mass of tissue. For small organisms, the amount of overestimation depends on the energy and yield of the photon. The photon contribution may be much smaller than the dose from electrons or alpha particles, and thus there is little effect on total dose. The DCFs are those of Amiro (1997), which were calculated by summing the doses from all radiations listed in ICRP (1983) for each radionuclide. All radionuclides with half-lives greater than 1 day are explicitly included, whereas those radionuclides with half-lives of less than 1 day are implicitly included by including their DCF values with that of the parent.

For external exposure, the DCFs of Holford (1989) were used. These are DCF values for humans from various exposure pathways using the EDEFIS computer code (Barnard and D'Arcy, 1986) with input data consistent with ICRP (1983). The model for water immersion (and air) is similar to that of Kocher (1983), but with slightly different exposure geometries. For water and soil/sediment immersion, the receptor is submerged 0.1 m below the surface of a semi-infinite, uniformly contaminated body of water or soil/sediment. Holford's (1989) DCF values for photons at the body surface and electrons at 70 µm into the skin were used. This assumes that most organisms have an epidermal layer that partially shields the body from electron radiation, but allows for penetration of photon energy.

3.4.1.8 Relative biological effectiveness (RBE) of alpha emitters and tritium

The extent of damage produced by an exposure to radiation is significantly affected by the RBE of the various types of radiation. The most important factor affecting the RBE is the LET, which is defined as the amount of energy dissipated by an ionizing particle per micrometre of path. For the purpose of human radiation protection, where the endpoint of greater concern is cancer, ICRP (1979) recommended that a weighting factor of 20 be used to account for the greater RBE of alpha particles and that a weighting factor of 1 be used for beta particles and gamma rays.

Studies conducted on non-human species for a variety of endpoints have reported increased biological effectiveness of alpha particles (range from 1 to over 300) and of ^{3}H (range between 1.8 and 3.8). Many of the radionuclides to which organisms are exposed in the Canadian environment include alpha emitters of the U decay series. In addition, more than 60% of the radiation dose to aquatic organisms exposed to routine discharges from Canadian nuclear power plants is from the beta emitter ^{3}H. There is, therefore, a need to incorporate relevant RBE factors for ^{3}H and alpha emitters in the DCFs used to assess doses of radiation to organisms in the Canadian environment.

NCRP (1991) stated that for non-human biota, weighting factors are required to modify the calculated absorbed dose and thus give a measure of the biologically effective dose in aquatic organisms. Based on the fact that the limited data available indicate that the RBE of various radiation

types in aquatic organisms is similar to that found in mammals (Woodhead, 1984), NCRP (1991) suggested that it would be reasonable to apply a weighting factor of 20 to the absorbed dose from alpha particles. Blaylock *et al*. (1993) also suggested that a weighting factor of 20 could be used when calculating doses to aquatic organisms from alpha particles. UNSCEAR (1996) has taken a different view, stating that the experimental data for animals indicate that a lower weighting factor of 5 for alpha radiation would be more appropriate and that the weighting factors for beta and gamma radiation should remain unity.

A careful review of the data presented in UNSCEAR (1996) did not reveal any data supporting the choice of a weighting factor of 5 for biota dose estimates in the context of ecological risk assessments. The only information dealing with the RBE of alpha emitters is the data on the reduction of primary oocyte survival in mice injected with a saline solution of polonium trichloride ($PoCl_3$). In this case, the RBE for the alpha-emitting ^{210}Po was estimated to be approximately 370. UNSCEAR (1996) also presented data on the RBE of ^{3}H relative to external gamma radiation (^{60}Co or ^{137}Cs) in the range of 1.8–3.8.

Straume and Carsten (1993) reviewed a large number of studies focusing on the RBE of ^{3}H beta rays. Their review indicates that RBEs for ^{3}H as tritium oxide (HTO) range from ~1 to 2.9, depending on the endpoint and cell system or species tested (Table 18). Based on these data and the data in UNSCEAR (1996), a weighting factor of 3 is recommended for this ecological risk assessment. This is consistent with Pentreath (1998), who recommended use of an RBE weighting factor of 3 for ^{3}H. This value is not considered overly conservative, because there are limited data on the RBEs of organically bound ^{3}H and because RBEs between 3 and 4 have been measured when ^{3}H is bound to amino acids or to nuclear bases such as thymidine (Straume and Carsten, 1993).

In the case of alpha emitters, a weighting factor of 40 is recommended for this ecological risk assessment. There is evidence that the damage caused by alpha radiation is fundamentally different from that of low-LET radiation (Goodhead *et al*., 1993). Results from several studies indicate unique types of damage, particularly to hematopoietic stem cells (Kadhim *et al*., 1992) and for sister chromatid exchange in lymphocytes (cited in Prestwich *et al*., 1995), from alpha particle exposure. The theoretical basis for this unique damage is the presence of large clusters of physical damage from alpha particle impact, which the cell cannot repair. These effects are not observed under exposure to low-LET radiation and hence indicate an RBE approaching infinity.

A large number of studies have assessed the RBE of high-LET radiation. In general, these studies have shown that RBE values are higher at low doses (i.e., environmentally relevant doses), because the effectiveness of low-LET radiation is more strongly dependent on the dose, dose rate and cellular conditions. Relative biological effectiveness values obtained from *in vitro* cell culture studies under uniform exposures for both low-LET and high-LET radiation tend to range from 1 to 10. *In vivo* studies have also been conducted to estimate the RBE values for high-LET alpha emitters. These *in*

vivo studies conducted at relatively low doses and dose rates are much closer to natural exposure conditions than *in vitro* studies that used very high doses and dose rates.

Studies with injected alpha emitters (^{210}Po, ^{239}Pu) in mice at low dose rates using such endpoints as oocyte mortality, sperm head abnormality and reduced immune function show much higher (>100) RBE values (e.g., Samuels, 1966; Rao *et al*., 1991; Jiang *et al*., 1994; Lord and Mason, 1996). These high RBE values were obtained using endpoints (reproduction and immune system function) that are important for the maintenance of healthy, actively reproducing populations.

In determining an appropriate weighting factor for the RBE of alpha emitters for use in estimating doses to biota, consideration should be given to the *in vivo* studies referred to above and to the dose estimates made during these studies. Accurate dose estimates are necessary for accurate determination of RBEs. The *in vivo* mice studies were conducted by injecting solutions of alpha emitters into the animals. Radiation doses from these injected alpha emitters were estimated assuming a uniform distribution of the radionuclide, which may not reflect the real dose to the critical target. The uncertainty in the dose estimates (i.e., underestimation of dose to the critical target) in the *in vivo* studies suggests that the true RBEs may be lower than those reported. Therefore, as a result of this uncertainty in the high RBE values and because of the relevance of the endpoints to population fitness, a weighting factor of 40 is recommended for use in this assessment for calculating doses to biota from alpha emitters.

3.4.2 Radiation exposure characterization and risk analysis

3.4.2.1 Uranium mines and mills

This section of the report focuses on the effect of radiation on aquatic organisms. The radiation dose is also calculated for terrestrial organisms — i.e., small mammals (voles), soil and litter invertebrates and plants (blueberry bushes, Labrador tea and lichen) — where data are available. Emphasis is placed on the aquatic environment, because it is the aquatic system that is most impacted by mining operations. Aquatic systems receive effluent discharges from TMAs/WMFs, groundwater discharges (dewatering effluents) from the ore bodies and ore milling effluents, and other waste discharges from mine facilities. Other work (MacLaren Plansearch Inc., 1987; SENES Consultants Ltd., 1996) has demonstrated that radiation effects on terrestrial biota are minor. The calculated radiation doses to terrestrial plants and animals near U mines and mills include the radiation dose from both inhalation and ingestion pathways. A description of both the operating and decommissioned U mines and mills evaluated in this assessment is given in Section 3.3.2.2.

Table 19 presents the realistic RQs calculated for exposure to radiation doses at various operating and decommissioned U mines. Risk quotients are also given for maximum radiation exposure under pre-operational conditions at the Cigar Lake, McClean Lake, Midwest Joint Venture and McArthur River mine areas before ore extraction and/or milling of the ore commenced, for comparative purposes and for the purpose of assessing the relevance of the ENEVs derived for this assessment.

Potential harmful effects occur at each of the mine sites, with RQs greater than 1 being calculated. Areas of potential effects are generally fairly localized and confined to water bodies receiving direct effluent inputs from mining/milling operations or TMAs/WMFs. An exception is the Beaverlodge Lake area, which appears to be more widely contaminated. Many of the data from the Beaverlodge Lake area are relatively old, especially for radionuclide concentrations in sediment and biota, and incomplete in terms of daughter radionuclide concentrations. For this reason, effort was not put into better defining the area of degradation at the Beaverlodge Lake sites, although these calculations clearly illustrate potential radiation dose effects on aquatic biota.

High RQs were also calculated for the Rabbit Lake area. These values are for the Link Lakes (and Horseshoe Lake), which were contaminated by past management practices — i.e., minewater was pumped into Upper Link Lake, and surface water was diverted to the lake along a drainage ditch until 1977. Sediment radionuclide concentrations are particularly high in these lakes, especially near the point of inflow.

Overall, there is a greater potential for radiation toxicity to benthic invertebrates than other aquatic invertebrates on the basis of the higher number of RQs greater than 1 from the sediment pathway than for pelagic species. This is reasonable, because ^{238}U and ^{232}Th decay chain radionuclides tend to be particle reactive and partition from the water to bottom sediments; as a result, sediment radionuclide concentrations are much higher than those in water.

The RQ greater than 1 for fish reported for maximum pre-operational conditions is from the Midwest Joint Venture mine area.

In summary, potentially harmful effects on biota may occur in localized areas in the near field that receive mine/mill effluents and tailings effluents from the U mines/mills. Outside of the near-field receiving environment, radionuclide concentrations do not generally have the potential of causing harmful effects.

Concentrations of ^{222}Rn and radionuclides associated with particulates in air are measured at a number of sites at each of the U mines. These sites include areas representing background conditions and areas affected by mine operations, including the mill grinder, shafts and TMAs. For the calculation of RQs from inhalation of ^{222}Rn by a small mammal — a mouse or vole — the maximum ^{222}Rn concentrations reported at a given site were used along with the maximum concentrations reported for the most recent year for which data are available, usually 1997. These values are given in Table 20.

A crude (screening-level) estimation of the radiation dose to a mouse from ^{222}Rn was obtained by assuming that a mouse could be reasonably well represented by a small (20-g) human, because a DCF for a mouse was not readily available. In these calculations, the ^{222}Rn concentration in air ($Bq \cdot m^{-3}$) was multiplied by the DCF (annual effective dose for humans from ^{222}Rn and daughters) of 2.64×10^{-4}

$mSv \cdot Bq^{-1} \cdot m^{-3}$ for inhalation of ^{222}Rn by humans (Duport, 1999) and corrected for body size (metabolic rate) by multiplying by a correction factor. Macdonald (1996) gives a body weight of 20 g ww for a mouse, and this value was used in the present calculations. The allometric equation

$$I = (0.54576(bw)^{0.8})/bw$$

for mammals was used to estimate the inhalation rate of a mouse, where I is the inhalation rate (m^3 $air \cdot kg^{-1}$ $bw \cdot d^{-1}$) and bw is body weight in kg live weight. Using this equation, an inhalation rate of 1.19 $m^3 \cdot kg^{-1}$ $bw \cdot d^{-1}$ was calculated for a 20-g mouse. A human inhalation rate of 0.33 $m^3 \cdot kg^{-1}$ $bw \cdot d^{-1}$ is based on ICRP's reference man (ICRP, 1975). The ratio of these two inhalation rates multiplied by 20 gave a correction factor of 72.12. This value accounts for the difference in metabolic rate (inhalation rate) between a mouse and a human. It also accounts for the use of an RBE of 40, instead of 20, for the effect of alpha radiation on non-human biota in the DCF of Duport (1999).

The radiation dose to a mouse from inhalation of ^{222}Rn and RQs are given in Table 20. The RQs were generally less than 1 and therefore not harmful. The exception was ^{222}Rn emissions from the mine exhaust location at the Midwest Joint Venture Project. The RQ for this location was 1.3, indicating that ^{222}Rn emissions can potentially have effects. The Midwest Joint Venture Project has not yet sought approval for further development. Therefore, ^{222}Rn emissions from these U mines and waste management facility are not likely to be harmful to small mammals.

3.4.2.2 Uranium refineries and conversion facilities and waste management areas

Uranium refineries and conversion facilities and WMFs have been described in Section 3.3.2.3. Realistic RQs calculated for U refineries and WMFs are presented in Table 21. Releases from the Blind River facility do not appear to be harmful to non-human organisms, but concentrations of radionuclides in Port Hope Harbour due to historic releases from the Port Hope U refinery are potentially harmful, especially to benthic invertebrates. Because current liquid releases from the Port Hope refinery are similar to those of the Blind River refinery and result in similar aqueous concentrations of U, current radionuclide releases from the Port Hope refinery are considered unlikely to cause environmental harm. The RQ for fish is higher for Port Hope than for Blind River (Table 21) as a result of historic releases.

Radionuclide releases from the Port Granby and Welcome WMFs (Table 21) do not appear to be causing environmental harm. In the case of the CRL WMF, the limited data available for East Swamp Stream indicate that releases are potentially harmful to fish. In East Swamp Stream, Port Granby and Welcome, limited data were available with which to assess the effects on the aquatic environment. Also, data were not available for radionuclide concentrations in sediment at most sites, so the impact of contaminated sediment on benthic invertebrates was not assessed at most of these facilities. Data presented in AECL (1999) also indicated potential harmful effects to mammals (the muskrat) and benthic invertebrates in Duke Swamp.

3.4.2.3 Nuclear generating stations

Ontario Power Generation (OPG) operates NGSs at three sites in Ontario: Pickering, Bruce and Darlington. The Pickering site is located near Pickering on Lake Ontario and has two NGSs, Pickering NGS-A and Pickering NGS-B. Each station consists of four nuclear reactors (units) and has electrical power ratings of 2060 and 2064 MWe, respectively. The Bruce site is near Kincardine on Lake Huron and also consists of two NGSs, Bruce NGS-A and Bruce NGS-B. Each generating station consists of four nuclear reactors and has electrical power ratings of 3076 and 3440 MWe, respectively. At the Bruce site, there is also a Radioactive Waste Management Operations Site (RWOS), which consists of a Waste Volume Reduction Facility and a Central Maintenance Facility (CMF). The RWOS facility receives waste from OPG's three sites and processes the nuclear waste for storage. The CMF provides radioactive laundering services for the three sites. The Darlington site is near Bowmanville on Lake Ontario and consists of one NGF and a Tritium Removal Facility. The Darlington NGS has four reactors with a total electrical power rating of 3524 MWe. The Tritium Removal Facility extracts ^{3}H from the tritiated heavy water and stores it on-site (LaMarre, 1998). Hydro-Québec operates a nuclear facility at Gentilly near Trois-Rivières, Quebec, on the St. Lawrence River. This NGS has one reactor with an electrical power rating of 675 MWe. A Solid Radioactive Waste Management Facility and a dry irradiated fuel storage facility are also on-site. New Brunswick Power Corporation operates a nuclear facility at Point Lepreau near Saint John, New Brunswick, in the Bay of Fundy. Point Lepreau has one reactor with an electrical power rating of 680 MWe and a Solid Radioactive Waste Management Facility.

All NGSs monitor radiological emissions in the vicinity of their stations and at background stations. This includes boundary locations to assess the potential dose to human population centres, with distance from the site and background locations well away from nuclear sources. The monitoring program and associated sampling locations for OPG's, Gentilly and Point Lepreau NGSs are reviewed by Richardson (1999). All NGSs have radiological monitoring programs that monitor gamma exposure, air, precipitation, milk, drinking water, aquatic and terrestrial environments, and groundwater/surface water.

Risk quotients to aquatic biota are presented in Table 22. These are RQs based on effluent concentrations, with the exception of the values for fish and algae at Point Lepreau, which are realistic and therefore should greatly overestimate the risk of effect to biota. In all cases, RQs are less than 1, and radionuclide releases for these NGSs can be considered unlikely to be harmful to aquatic life.

3.5 Abiotic atmospheric effects

Radionuclides released to the atmosphere in particulate form are likely to be removed by particle scavenging and subsequently by wet and dry precipitation. This applies for all radionuclides, except ^{3}H, ^{14}C and the noble gases, such as ^{222}Rn. Radon-222 has a short half-life, 3.82 days, and decays to ^{210}Pb, which has a half-life of 22.3 years. Lead-210 is removed from the atmosphere by deposition

processes. Tritium is present in the atmosphere primarily as water vapour (HTO) or as hydrogen gas (HT), while ^{14}C is present primarily as carbon dioxide ($^{14}CO_2$). Ozone-depleting substances usually contain chlorine or bromine atoms, a characteristic that does not apply to the radionuclides released from nuclear facilities. Both the Photochemical Ozone Creation Potential and Global Warming Potential of the radionuclides released are low, since the quantities of radioactive carbon and hydrogen are very small relative to the release of stable forms from other sources.

4.0 PROPOSED CONCLUSIONS OF ASSESSMENT OF "TOXIC" UNDER CEPA 1999

CEPA 64(a): Based on available data concerning the effects (from exposure to both uranium and ionizing radiation) of releases of radionuclides from uranium mines and mills and waste management areas, it has been proposed that these releases are entering the environment in quantities or concentrations or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity.

Based on available data concerning the effects (from exposure to both uranium and ionizing radiation) of releases of radionuclides from uranium refineries and conversion facilities and power and research reactors, it has been proposed that these releases are not entering the environment in quantities or concentrations or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity.

CEPA 64(b): Based on available data, it has been proposed that releases of radionuclides from nuclear facilities are not entering the environment in quantities or concentrations or under conditions that constitute or may constitute a danger to the environment on which life depends.

Proposed overall conclusion: Based on critical assessment of relevant information, it has been proposed that releases of radionuclides from uranium mines and mills and waste management facilities are considered to be "toxic" as defined in Section 64 of CEPA 1999.

4.1 Considerations for follow-up (further action)

Since it is proposed that radionuclide releases from U mines and mills and WMFs be considered "toxic" as defined in Section 64 of CEPA 1999, it is recommended that investigations of options to reduce exposure to releases of radionuclides from U mines and mills and WMFs be considered a high priority. Discussions have been initiated with the CNSC to determine whether it will be possible to manage these releases under the new *Nuclear Safety and Control Act*. It is proposed that the process for risk management be formalized in the memorandum of understanding currently being negotiated between Environment Canada and the CNSC. It is recommended that priority be given to the establishment of both U and radionuclide water quality and sediment quality guidelines, if this is possible, based on the data available. Further research into the ENEVs for exposure of non-human biota to radiation should be a priority. This should include the genetic effects of environmentally relevant radiation doses and research into the effects of alpha emitters on ecologically relevant endpoints, for the purpose of verifying the appropriateness of the weighting factor to account for the greater RBE of alpha emitters. In PSL2 assessments, data on genetic damage were not taken into consideration in the derivation of estimated-no-effects values because of the difficulty in interpreting the significance of these effects at the population

level (i.e. population fitness and survival). Therefore, priority should also be given to research on the ecological significance of genetic effects and their consideration in the ecological risk assessment of radioactive and non-radioactive environmental contaminants. The information contained in this report could be utilized in making management decisions concerning the protection of the environment from radionuclide releases from other industries (non-nuclear), as well as for making risk management decisions pertaining to unlicensed, closed or abandoned U mines.

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GLOSSARY

Acute toxicity test — a toxicity test of short duration in relation to the life span of the test organism (e.g., usually =4 days for fish).

Allometric equation — equation based on the growth of the organism or the organism's energy requirements.

Alpha radiation — the emission of alpha particles from the nucleus of an unstable atom (radionuclide). Since particles transfer their energy in a very short distance and cannot penetrate the outer layer of skin, alpha radiation is an internal radiation hazard.

Application factor — a value by which the Critical Toxicity Value is divided to give the Estimated No-Effects Value.

Becquerel (Bq) — the SI unit of radioactivity for measuring the rate of decay of a radioactive substance. It is equivalent to the disintegration of one radioactive nucleus per second.

Benthic invertebrate — aquatic invertebrate living on or in the sediment.

Benthos — synonym for benthic invertebrate.

Beta radiation — the emission of electrons or positrons from the nucleus of an unstable atom (radionuclide). Beta particles can penetrate biological tissue to a depth of 1–2 cm. They may pose both an internal and an external hazard.

Biokinetics — the uptake, transport and distribution of a substance within an organism after ingestion.

CANDU — Canadian Deuterium Uranium, the name of the Canadian-designed reactor that uses natural uranium fuel and is moderated by heavy water. CANDU is a registered trademark of Atomic Energy of Canada Limited.

Chronic toxicity test — a toxicity test that spans a significant portion of the life span of the test organism (e.g., 10% or more) and examines effects on such parameters as metabolism, growth, reproduction and survival.

Concentration ratio (CR) — the ratio of the steady-state concentration of a substance in an organism due to uptake via contact with water to the concentration of the substance in the test water; and/or the ratio of the uptake rate constant to the depuration constant, assuming first-order kinetics.

Conservative — a cautious estimate that overestimates the dose to biota.

Critical Toxicity Value (CTV) — the quantitative expression (e.g., EC_{10}) of low toxic effect to the measurement endpoint. CTVs are used in risk characterization for the calculation of an Estimated No-Effects Value.

Daughter — any nuclide that originates from another nuclide by radioactive decay.

EC_x — the concentration of a substance that is estimated to have a specified effect (e.g., immobilization, reduced growth) on x% of the test organisms. The duration of the test must be specified.

Gamma radiation — the emission of photons (gamma rays), which carry energy but no charge, by an unstable atom (radionuclide). Gamma radiation is the most highly penetrating radiation.

Gray (Gy) — the SI unit of absorbed dose for ionizing radiation, equal to 1 joule of radiation energy absorbed in 1 kilogram of the material of interest.

Half-life — the time required for 50% of the activity of a given radionuclide to decay.

LC_{50} — the concentration of a substance that is estimated to be lethal to 50% of the test organisms over a specified period of time.

LD_{50} — the dose that causes mortality in 50% of the organisms tested.

Linear energy transfer (LET) — the rate of energy loss per unit path length.

Lowest Effect Level — the concentration at which actual ecotoxic effects become apparent; the 5th percentile of the screening-level concentration.

Lowest-Observed-Adverse-Effect Level (LOAEL) — the lowest dose in a toxicity test that caused a statistically significant adverse effect in comparison with the controls.

Macrophyte — a member of macroscopic plant life, especially of a body of water.

No-Observed-Effect Level (NOEL) — the highest dose in a toxicity test not causing a statistically significant effect compared with the controls.

Partition coefficient (K_d) — a measure of the propensity of a particular radionuclide to associate with solid phases; defined as the ratio of the concentration of the radionuclide on the particulate fraction to the concentration in water ($L \cdot kg^{-1}$ dry sediment).

Pelagic biota — aquatic organisms living in the water column of a body of water, rather than along the shore or in the bottom sediments.

Photosynthesis — the elaboration of organic matter (carbohydrate) from carbon dioxide and water with the aid of light energy.

Phytoplankton — the plant component of plankton.

Radioactive decay — the changing and progressive decrease in the number of unstable atoms (radionuclides) in a substance, due to their spontaneous nuclear disintegration or transformation into different atoms, during which particles and/or photons are emitted.

Radiological dose — the strict definition of radiological dose is the energy absorbed per unit mass of biological tissue exposed to ionizing radiation, measured in grays (Gy).

Realistic — a more accurate estimate of the dose to biota.

Risk quotient — a measure of potential toxicity derived by dividing the estimated exposure value (e.g., total radiation dose) by the Estimated No-Effects Value.

Screening-level concentration (SLC) approach — an effects-based approach applicable to benthic invertebrates that uses field data on the co-occurrence in sediments of benthic infaunal species and different concentrations of contaminants to estimate the effect of a contaminant on benthic species.

Secular equilibrium — an equilibrium reached between a precursor and daughter nuclide in which the daughter nuclide decays at the same rate as it is produced. The daughter nuclide must be much shorter-lived than the precursor. Secular equilibrium is reached after a period equivalent to 6–10 daughter half-lives. See also "radioactive decay."

Severe Effect Level — the sediment concentration that could potentially eliminate most of the benthic invertebrates; defined as the 95th percentile of the screening-level concentration.

Sievert (Sv) — the SI unit of dose equivalent. 1 Sv equals 1 joule per kilogram. See also "radioactive decay."

Transfer coefficient (TC) — a ratio that combines the concept of the concentration ratio (unitless) with the rate of intake of the organism ($kg \cdot d^{-1}$) to predict the fraction of the radionuclide in the diet that is accumulated in the body on a daily basis ($d \cdot kg^{-1}$).

UNSCEAR — United Nations Scientific Committee on the Effects of Atomic Radiation established by the General Assembly in 1955. It reports annually to the General Assembly and submits comprehensive reports with scientific annexes at irregular intervals. These review reports that have had a strong influence on radiation protection standards, e.g., by the International Commission on Radiological Protection.

Zooplankton — the animal component of plankton.

Table 1: Summary of uranium toxicity studies reported in the literature

Method	Species	Endpoint	Dose	Reference
Dermal	Rabbit	LD_{50}	0.13 g U·kg^{-1} bw	Durbin and Wrenn (1975)
	Rat	LD_{50}	1 g U·kg^{-1} bw	Durbin and Wrenn (1975)
	Guinea pig	LD_{50}	4 g U·kg^{-1} bw	Durbin and Wrenn (1975)
	Mouse	LD_{50}	16 g U·kg^{-1} bw	Durbin and Wrenn (1975)
Injection	Rabbit	LD_{50}	0.1 mg U·kg^{-1} bw	Durbin and Wrenn (1975)
	Guinea pig	LD_{50}	0.3 mg U·kg^{-1} bw	Durbin and Wrenn (1975)
	Rat	LD_{50}	1–2 mg U·kg^{-1} bw	Durbin and Wrenn (1975)
	Dog	LD_{50}	2 mg U·kg^{-1} bw	Durbin and Wrenn (1975)
	Mouse	LD_{50}	20–25 mg U·kg^{-1} bw	Durbin and Wrenn (1975)
	Dog	>60% decrease in kidney filtration rate	0.5 mg U·kg^{-1} bw	Leggett (1989)
	Rat	>60% decrease in kidney filtration rate	10 mg U·kg^{-1} bw	Leggett (1989)
Long-term feeding	Rat	100% mortality	2–10% soluble U for 30 days	Yuile (1973)
	Rat	no excess mortality	20% insoluble U for 30 days	Yuile (1973)
	Rabbit	LD_{50}	23 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Dog	LD_{50}	47 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Rat (male)	LD_{50}	1070 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Mouse (C3H)	LD_{50}	2000 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Dog	kidney pathology	9.4 mg U·kg^{-1} bw·d^{-1}	Maynard and Hodge (1949)
Inhalation	Rabbit	LD_{50}	0.07 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Dog	LD_{50}	0.42 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Guinea pig	LD_{50}	1.7 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)
	Mouse (albino)	LD_{50}	1.7 mg U·kg^{-1} bw·d^{-1}	Durbin and Wrenn (1975)

Table 2: Acute toxicity of uranium to fish from the list given in Bird *et al.* (2000)

Species	Scientific name	Endpoint/life stage	Type of test	U conc. ($mg \cdot L^{-1}$)	Hardness ($mg \cdot L^{-1}$)	Alkalinity ($mg \cdot L^{-1}$)	Reference
fathead minnow	*Pimephales promelas*	mortality	96-h LC_{50}	2.8–3.1	20	–	Tarzwell and Henderson (1960)
brook trout	*Salvelinus fontinalis*	mortality/adult	96-h LC_{50}	5.5	35.1	11	Parkhurst *et al.* (1984)
rainbow trout	*Oncorhynchus mykiss*	mortality	96-h LC_{50}	6.2	31	–	Davies (1980)
brook trout	*Salvelinus fontinalis*	mortality	96-h LC_{50}	8.0	31	–	Davies (1980)
brook trout	*Salvelinus fontinalis*	mortality/adult	96-h LC_{50}	23	208	53	Parkhurst *et al.* (1984)
fathead minnow	*Pimephales promelas*	mortality	96-h LC_{50}	135	400	–	Tarzwell and Henderson (1960)

Table 3: Acute and chronic toxicity of uranium to invertebrates from the list given in Bird *et al*. (2000).

Species[1]	Site	Endpoint	U conc. ($mg \cdot L^{-1}$)	Hardness ($mg \cdot L^{-1}$)	Alkalinity ($mg \cdot L^{-1}$)	Reference
Ceriodaphnia dubia	reproduction/1-d instars	7-d chronic EC_{50}	0.003[2]	5	3	Pickett *et al*. (1993)
Ceriodaphnia dubia	mortality/1-d instars	48-h LC_{50}	0.073[2]	5	3	Pickett *et al*. (1993)
Hydra viridissima (hydra)	growth	LOAEL	0.15	–	–	Hyne *et al*. (1992)
4 cladoceran species	mortality/1-d instars	24-h LC_{50}	0.14–0.9	–	3.26	Bywater *et al*. (1991)
Daphnia pulex	mortality/1-d instars	48-h LC_{50}	0.22	3.1	0.3	Trapp (1986)
Daphnia magna	reproduction/1-d instars	long-term chronic test LOAEL	0.52	66–73	54–60	Poston *et al*. (1984)
Daphnia magna	mortality/1-d instars	48-h LC_{50}	74.3–29.6	188–205	124–133	Poston *et al*. (1984)

[1] Crustacean zooplankton except for hydra.
[2] Value is a mean of several tests conducted by two independent laboratories (Pickett *et al*., 1993).

Table 4: Summary of CTVs and ENEVs derived for uranium in wildlife and aquatic biota

Species	CTV	Application factor	ENEV
Terrestrial mammals and birds	0.31 $mg \cdot kg^{-1}$ $bw \cdot d^{-1}$	1	0.31 $mg \cdot kg^{-1}$ $bw \cdot d^{-1}$
Terrestrial plants	64 $mg \cdot kg^{-1}$ dw soil	1	64 $mg \cdot kg^{-1}$ dw soil
Soil invertebrates	1000 $mg \cdot kg^{-1}$ dw soil	10	100 $mg \cdot kg^{-1}$ dw soil
Fish	2.8 $mg \cdot L^{-1}$	10	280 $\mu g \cdot L^{-1}$
Daphnia pulex	22 $\mu g \cdot L^{-1}$	1	22 $\mu g \cdot L^{-1}$
Ceriodaphnia dubia	3 $\mu g \cdot L^{-1}$	1	3 $\mu g \cdot L^{-1}$
Benthic invertebrates	21 $mg \cdot kg^{-1}$ dw sediment	1	21 $mg \cdot kg^{-1}$ dw sediment

Table 5: Estimated background exposure for representative species in Saskatchewan[1]

Species	Body weight (g)	Water			Food				Soil/sediment			Total U intake ($\mu g \cdot d^{-1}$)	Dose ($\mu g \cdot g^{-1}$ bw$\cdot d^{-1}$)	ENEV ($\mu g \cdot g^{-1}$ bw$\cdot d^{-1}$)	Risk quotient
		Water intake rate ($L \cdot d^{-1}$)	U conc. in water ($\mu g \cdot L^{-1}$)	U intake ($\mu g \cdot d^{-1}$)	Food item	Total daily food intake ($g \cdot d^{-1}$)	U conc. in food ($\mu g \cdot g^{-1}$ ww)	U intake ($\mu g \cdot d^{-1}$)	Total daily soil intake ($g \cdot d^{-1}$)	U conc. in sed./soil ($\mu g \cdot g^{-1}$ ww)	U intake ($\mu g \cdot d^{-1}$)				
mallard	883	0.16	0.35	0.04	insects	26.2	1.970	52	0.9	8.86	8	75.8	0.09	0.31	0.28
					plants	45.9	0.062	3	1.5	8.86	13				
osprey	1725	0.26	0.35	0.06	sucker	321.2	0.100	32	6.4	8.86	57	88.9	0.05	0.31	0.17
red fox	4128	1.06	0.35	0.24	small mammals	565.1	0.008	5	15.8	0.91	14	19.1	0.005	0.31	0.02
mink	578	0.18	0.35	0.04	fish	135.7	0.100	14	2.7	8.86	24	37.5	0.06	0.31	0.21
muskrat	1300	0.38	0.35	0.09	macrophytes	212.6	0.062	13	10.6	8.86	94	107	0.08	0.31	0.27

[1] Data are from several sources. Where possible, data from northern Saskatchewan are used in preference to data from other sites.

Table 6: Potential effects of uranium in the Serpent River watershed/Elliot Lake area for wildlife, crustacean zooplankton, benthic invertebrates and fish based on uranium concentrations in water, sediment/soil and biota

	Hough Lake	Ten Mile	Dunlop Lake	McCabe Lake	Quirke Lake	Kindle Lake	Elliot Lake A	Elliot Lake B	Whiskey Lake
U concentrations (EEVs)									
water ($\mu g \cdot L^{-1}$)	1.7	0.5	0.5	7	15.3	8.1	3	0.7	7.2
sediment ($mg \cdot kg^{-1}$ dw)	89.5	13	39.3	164.6	76.2	46.7	282	282	78.3
fish ($mg \cdot kg^{-1}$ ww)	–	–	8.4	51	4.46	–	0.65	0.65	0.5
macrophyte ($mg \cdot kg^{-1}$ ww)	–	–	0.0095	1.45	–	–	–	–	–
soil ($mg \cdot kg^{-1}$ dw)	–	–	–	–	–	–	–	–	–
small mammals ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	–	–	–	–
Risk quotients (RQ)[1]									
mallard	0.22	0.03	0.40	0.69	0.33	0.19	0.67	0.65	0.25
osprey	0.28	0.04	**5.3**	**32**	**3.0**	0.15	**1.28**	**1.27**	0.56
mink	0.35	0.05	**6.7**	**41**	**3.8**	0.20	**1.61**	**1.61**	0.70
muskrat	0.61	0.09	0.27	**1.9**	0.55	0.33	**1.92**	**1.91**	0.54
Ceriodaphnia (zooplankton)	0.57	0.17	0.17	**2.3**	**5.1**	**2.7**	1.0	0.23	**2.4**
Daphnia (zooplankton)	0.08	0.02	0.02	0.32	0.70	0.37	0.14	0.03	0.33
benthic invertebrates	**4.3**	0.6	**1.9**	**8**	**3.6**	**2.2**	**13.4**	–	**3.7**
fish	0.01	0.002	0.002	0.03	0.05	0.03	0.01	0.003	0.03

[1] ENEVs were 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$ for wildlife, 3 $\mu g \cdot L^{-1}$ for *Ceriodaphnia*, 22 $\mu g \cdot L^{-1}$ for *Daphnia*, 21 $mg \cdot kg^{-1}$ dw sediment for benthic invertebrates and 280 $\mu g \cdot L^{-1}$ for fish. RQs greater than 1 are in boldface type.

Table 7: **Potential effects of uranium in the Beaverlodge Lake area for wildlife, crustacean zooplankton, benthic invertebrates and fish based on average uranium concentrations in water, sediment/soil and biota**

	Beaverlodge Lake	Dubyna Lake A	Dubyna Lake B	Ace Creek AC-14	Greer Lake[1]	Marie Lake
U concentration (EEVs)						
water ($\mu g \cdot L^{-1}$)	168	333	90	59	–	1155
sediment ($mg \cdot kg^{-1}$ dw)	397	454	454	37	2120	414
fish ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	–
macrophyte ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	–
soil ($mg \cdot kg^{-1}$ dw)	–	–	–	–	–	–
small mammal ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	–
Risk quotients (RQ)[2]						
mallard	**2.6**	**4.4**	**2.0**	0.69	**4.8**	**13**
osprey	**1.6**	**1.8**	**1.5**	0.19	**6.6**	**2.7**
red fox	**–**	**–**	**–**	**–**	**–**	**–**
mink	**2.1**	**2.4**	**1.9**	0.26	**8.3**	**3.9**
muskrat	**3.0**	**3.7**	**3.2**	0.36	**14.4**	**4.9**
Ceriodaphnia (zooplankton)	**56.0**	**111**	**30**	**20**	–	**385**
Daphnia (zooplankton)	**7.6**	**15**	**4.1**	**2.7**	–	**53**
benthic invertebrates	**19**	**22**	–	**1.8**	**70–164**	**11–36**
fish	0.6	**1.19**	0.32	0.21	–	**4.1**

[1] RQs based on the sediment pathway.

[2] ENEVs were 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$ for wildlife, 3 $\mu g \cdot L^{-1}$ for *Ceriodaphnia*, 22 $\mu g \cdot L^{-1}$ for *Daphnia*, 21 $mg \cdot kg^{-1}$ dw sediment for benthic invertebrates and 280 $\mu g \cdot L^{-1}$ for fish. RQs greater than 1 are in boldface type.

Table 8: Potential effects of uranium in the Cluff Lake Project area for wildlife, crustacean zooplankton, benthic invertebrates and fish based on whole-lake average uranium concentrations in water, sediment/soil and biota

	Island Lake	Snake Lake	Cluff Lake
U concentration (EEVs)			
water ($\mu g \cdot L^{-1}$)	197	5.6	0.18
sediment ($mg \cdot kg^{-1}$ dw)	701	25.5	42.3
fish ($mg \cdot kg^{-1}$ ww)	0.31	–	0.18
macrophyte ($mg \cdot kg^{-1}$ ww)	12.9	0.22	2.07
soil ($mg \cdot kg^{-1}$ dw)	19	19	19
small mammal ($mg \cdot kg^{-1}$ ww)	0.13	0.13	0.13
Risk quotients (RQ)[1]			
mallard	**5.2**	0.15	0.46
osprey	**1.7**	0.09	0.24
red fox	0.4	0.25	0.24
mink	**2.2**	0.11	0.31
muskrat	**10.4**	0.30	**1.41**
Ceriodaphnia (zooplankton)	**66**	**1.9**	0.06
Daphnia (zooplankton)	**9.0**	0.25	0.008
benthic invertebrates	**33**	**1.2**	**2**
fish	0.7	0.02	0.001

[1] ENEVs were 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$ for wildlife, 3 $\mu g \cdot L^{-1}$ for *Ceriodaphnia*, 22 $\mu g \cdot L^{-1}$ for *Daphnia*, 21 $mg \cdot kg^{-1}$ dw sediment for benthic invertebrates and 280 $\mu g \cdot L^{-1}$ for fish. RQs greater than 1 are in boldface type.

Table 9: Potential effects of uranium in the Rabbit Lake Project area for wildlife, crustacean zooplankton, benthic invertebrates and fish based on average uranium concentrations in water, sediment/soil and biota

	Link Lakes				Pow Wow Bay	Hidden Bay	Horseshoe Lake[1]	Collins Bay Eagle Point	Regional basin maxima[1]
	SO – Airport Road[1]	SI – Sedimentation Dam 1984[1]	SI – Sedimentation Dam 1985[1]	Upper Link Lakes 1980[1]					
U concentration (EEVs)									
water ($\mu g \cdot L^{-1}$)	–	–	–	–	0.52	0.7	–	1.21	–
sediment ($mg \cdot kg^{-1}$ dw)	3600	4000	1690	5650	11.7	35.2	2040	25.3	520
fish ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	0.006	–	0.004	–
macrophyte ($mg \cdot kg^{-1}$ ww)	–	–	–	–	0.42	1.88	–	0.46	–
soil ($mg \cdot kg^{-1}$ dw)	–	–	–	–	–	–	–	–	–
small mammal ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	–	–	–	–
Risk quotient (RQ)[2]									
mallard	**8.2**	**9.1**	**4.3**	**13**	0.10	0.41	**4.6**	0.14	**1.2**
osprey	**11.1**	**12.4**	**6.1**	**18**	0.04	0.11	**6.3**	0.07	**1.6**
red fox	**–**	**–**	**–**	**–**	–	–	**–**	–	**–**
mink	**14.0**	**16**	**8**	**22**	0.05	0.14	**8**	0.08	**2.0**

	Link Lakes				Pow Wow Bay	Hidden Bay	Horseshoe Lake[1]	Collins Bay Eagle Point	Regional basin maxima[1]
	SO – Airport Road[1]	SI – Sedimentation Dam 1984[1]	SI – Sedimentation Dam 1985[1]	Upper Link Lakes 1980[1]					
muskrat	**25**	**27.2**	**13.3**	**38**	0.31	**1.26**	**14**	0.39	**3.5**
Ceriodaphnia (zooplankton)	**–**	**–**	**–**	**–**	0.17	0.23	**–**	0.4	**–**
Daphnia (zooplankton)	**–**	**–**	**–**	**–**	0.02	0.03	**–**	0.06	**–**
benthic invertebrates	**171**	**191**	**93**	**269**	0.6	**1.7**	**97**	1	**25**
fish	–	–	–	–	0.002	0.003	–	0.004	–

[1] RQs based on sediment U concentrations only.

[2] ENEVs were 0.31 mg U·kg^{-1} bw·d^{-1} for wildlife, 3 µg·L^{-1} for *Ceriodaphnia*, 22 µg·L^{-1} for *Daphnia*, 21 mg·kg^{-1} dw sediment for benthic invertebrates and 280 µg·L^{-1} for fish. RQs greater than 1 are in boldface type.

Table 10: Potential effects of uranium in the Key Lake Project area for wildlife, crustacean zooplankton, benthic invertebrates and fish based on average uranium concentrations in water, sediment/soil and biota

	Horsefly Lake[1]	McDonald Lake	Little McDonald[1]	Delta	David Creek	Maximum baseline
U concentration (EEVs)						
water ($\mu g \cdot L^{-1}$)	–	1.7	–	2.2	2	5.4
sediment ($mg \cdot kg^{-1}$ dw)	3020	171	472	4.8	212	12
fish ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	0.003
macrophyte ($mg \cdot kg^{-1}$ ww)	–	0.15	–	–	0.18	0.46
soil ($mg \cdot kg^{-1}$ dw)	45	45	45	45	45	2.1
small mammal ($mg \cdot kg^{-1}$ ww)	0.065	0.065	0.065	0.065	0.065	0.065
Risk quotients (RQ)[2]						
mallard	**7**	0.4	**1.1**	0.03	0.5	0.2
osprey	**9**	0.8	**1.6**	0.02	0.07	0.0
red fox	0.6	0.6	0.6	0.61	0.6	0.0
mink	**12**	**1.1**	**1.8**	0.02	0.8	0.0
muskrat	**21**	**1.2**	**3.2**	0.04	**1.5**	0.3
Ceriodaphnia (zooplankton)	**–**	0.57	**–**	0.73	0.67	**1.8**
Daphnia (zooplankton)	**–**	0.08	**–**	0.10	0.09	0.2
benthic invertebrates	**144**	**8.1**	**22**	0.2	**10**	0.6
fish	–	0.01	–	0.01	0.01	0.01

[1] RQs based on the sediment pathway.

[2] ENEVs were 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$ for wildlife, 3 $\mu g \cdot L^{-1}$ for *Ceriodaphnia*, 22 $\mu g \cdot L^{-1}$ for *Daphnia*, 21 $mg \cdot kg^{-1}$ dw sediment for benthic invertebrates and 280 $\mu g \cdot L^{-1}$ for fish. RQs greater than 1 are in boldface type.

Table 11: Potential effects of uranium on wildlife, crustacean zooplankton, benthic invertebrates and fish at maximum baseline concentrations measured in water, sediment, fish, macrophytes, soil and small mammals at Cigar Lake, McClean Lake, Midwest Joint Venture and McArthur River areas

Maximum U concentrations (EEVs)		**Tier risk quotient (RQ)[1]**	
water ($\mu g \cdot L^{-1}$)	5.4	mallard	0.16
sediment ($mg \cdot kg^{-1}$ dw)	3.5	osprey	0.04
fish ($mg \cdot kg^{-1}$ ww)	0.003	red fox	0.04
macrophyte ($mg \cdot kg^{-1}$ ww)	0.46	mink	0.06
soil ($mg \cdot kg^{-1}$ dw)	2.1	muskrat	0.34
small mammals ($mg \cdot kg^{-1}$ ww)	0.01	*Ceriodaphnia* (zooplankton)	**1.8**
		Daphnia (zooplankton)	0.25
		benthic invertebrates	**1.7**
		fish	0.02

[1] ENEVs were 0.31 mg $U \cdot kg^{-1}$ $bw \cdot d^{-1}$ for wildlife, 3 $\mu g \cdot L^{-1}$ for *Ceriodaphnia*, 22 $\mu g \cdot L^{-1}$ for *Daphnia*, 21 $mg \cdot kg^{-1}$ dw for benthic invertebrates and 280 $\mu g \cdot L^{-1}$ for fish. RQs greater than 1 are in boldface type.

Table 12: Potential effects of uranium at uranium refineries and waste management areas for wildlife, crustacean zooplankton, benthic invertebrates and fish based on average uranium concentrations in water, sediment/soil and biota

	Blind River[1]	Port Hope Harbour	Port Granby A[1,2]	Port Granby B[1,2]	Port Granby C[1,2]	Welcome A[1]	Welcome B[1]
U concentration (EEVs)							
water ($\mu g \cdot L^{-1}$)	0.6	10	900	100	170	21	85
sediment ($mg \cdot kg^{-1}$ dw)	–	1280	–	–	–	–	–
fish ($mg \cdot kg^{-1}$ ww)	–	0.2	–	–	–	–	–
macrophyte ($mg \cdot kg^{-1}$ ww)	–	8.12	–	–	–	–	–
soil ($mg \cdot kg^{-1}$ dw)	2.57	–	45	45	45	44	44
small mammal ($mg \cdot kg^{-1}$ ww)	–	–	–	–	–	–	–
Risk quotients (RQ)[3]							
mallard	0.01	**4.4**	**9.2**	0.1	**1.7**	0.22	0.87
osprey	0.00	**4.0**	**1.1**	0.01	0.22	0.03	0.11
red fox	0.04	0.74	**1.3**	0.58	0.72	0.58	0.64
mink	0.00	**5.0**	**1.8**	0.02	0.34	0.04	0.17
muskrat	0.00	**13.1**	**1.6**	0.02	0.30	0.04	0.15
Ceriodaphnia (zooplankton)	0.20	**3.3**	**300**	**3.3**	**57**	**7**	**28.3**

	Blind River[1]	Port Hope Harbour	Port Granby A[1,2]	Port Granby B[1,2]	Port Granby C[1,2]	Welcome A[1]	Welcome B[1]
Daphnia (zooplankton)	0.03	0.45	**40.9**	0.45	**7.7**	0.95	**3.86**
benthic invertebrates	–	**61**	–	–	–	–	–
fish	0.002	0.04	**3.2**	0.04	0.04	0.08	0.30

[1] RQs do not include U from sediment.

[2] Port Granby A is the entrance to overflow culvert at East Reservoir, while Port Granby B is the point of entry of water into Lake Ontario on January 24, 1993. Port Granby C is 10 m west of the entry point to Lake Ontario on January 5, 1993.

[3] ENEVs were 0.31 mg U·kg^{-1} bw·d^{-1} for wildlife, 3 µg·L^{-1} for *Ceriodaphnia*, 22 µg·L^{-1} for *Daphnia*, 21 mg·kg^{-1}dw for benthic invertebrates and 280 µg·L^{-1} for fish. RQs greater than 1 are in boldface type.

Table 13: Potential effects of recent maximum uranium concentrations measured in soil near the Port Hope uranium refinery

	Maximum U concentration ($mg \cdot kg^{-1}$ dw)	
	52[1]	90[2]
Terrestrial plant realistic RQ	0.81	**1.41**
Soil invertebrate conservative RQ	0.52	0.90

[1] Low-level Radioactive Waste Management Office (1995).
[2] Kinch and McLaughlin (1998). RQs greater than 1 are in boldface type.

Table 14: Lower limits of effects as a result of radiation exposure reported in international reviews[1]

Source	Endpoint	Dose rate	Mammals	Birds	Amphibians	Reptiles
UNSCEAR (1996)	death	acute	1.6–16 Gy	5 Gy	adults: 2–22 Gy tadpoles: 0.1 Gy	2–22 Gy
	reproduction	chronic	0.9 $Gy \cdot a^{-1}$	0.05 $Gy \cdot a^{-1}$	2 $Gy \cdot a^{-1}$	2 $Gy \cdot a^{-1}$
	embryotoxicity	chronic	0.9 $Gy \cdot a^{-1}$	70 $Gy \cdot a^{-1}$	–	–
IAEA (1992)	death ($LD_{50/30}$)	acute	0.1 Gy	0.1 Gy	0.1 Gy	0.1 Gy
	reproduction	chronic	0.4 $Gy \cdot a^{-1}$	0.4 $Gy \cdot a^{-1}$	0.4 $Gy \cdot a^{-1}$	0.4 $Gy \cdot a^{-1}$
Rose (1992)	all effects	chronic	1 $Gy \cdot a^{-1}$	1 $Gy \cdot a^{-1}$	1 $Gy \cdot a^{-1}$	1 $Gy \cdot a^{-1}$

[1] Values are either the lowest reported values demonstrating an effect or limits at which the authors suggest that the effect will not be observed.

Table 15: Effects of acute radiation doses on survival of early life stages of fish expressed as an LD_{50} in Gy

LD_{50}[1]	**Species**	**Radiation source**	**Reference**
0.16 (150 days)	*Oncorhynchus kisutch* (1-cell stage)	X-ray	Bonham and Welander (1963)
0.58 (55 days)	*Salmo gairdneri* (1-cell stage)	X-ray	Welander (1954)
0.9	*Pleuronectes platessa* (embryos)	X-ray	Ward *et al*. (1971)
3.0	*Salmo gairdneri* (32-cell stage)	X-ray	Welander (1954)
5.0	*Salmo gairdneri* (late eye embryo)	X-ray	Welander (1954)
11.2 (30 days)	*Fundulus heteroclitus* (post-larvae)	^{60}Co	White and Angelovic (1966)
25 (60 days)	*Oncorhynchus tshawytscha* (embryos)	X-ray	Welander *et al*. (1948)

[1] The number of days in parentheses is the number of days after exposure at which the LD_{50} was determined.

Table 16: Summary of ENEVs used to assess the potential toxicity of exposure of non-human biota to radiation near Canadian nuclear facilities

Taxa	**Realistic ENEV ($Gy \cdot a^{-1}$)**
fish	0.2
benthic invertebrates	0.6
algae	0.88
macrophytes	0.88
amphibians	0.08
small mammals	0.4
terrestrial plants	0.88
terrestrial invertebrates	0.88

Table 17: Maximum and geometric mean (GM) concentration ratios ($L \cdot kg^{-1}$ ww) between water and biota used to predict radionuclide concentrations in biota from aqueous radionuclide concentrations

	Fish		Algae		Macrophytes	
	GM	**Maximum**	**GM**	**Maximum**	**GM**	**Maximum**
^{3}H	1	1	1	1	1	1
^{60}Co	14	650	950	30 000	790	15 000
^{90}Sr	240	8000	135	600	135	2167
^{134}Cs	76	26 432	16	2672	49	21 200
^{137}Cs	76	26 432	16	2672	49	21 200
^{144}Ce	100	1000	100	1000	100	1000
^{106}Ru	3	30	755	10 000	100	1000
^{230}Th	9	30	250	36 000	500	72 000
^{226}Ra	6	196	755	3500	1560	6900
^{222}Rn	1	1	1	1	1	1
^{210}Pb	8	140	75	10 800	100	1400
^{210}Bi	15	2160	3.75	540	7.5	2160
^{210}Po	42	166	125	18 000	250	36 000
^{228}Th	9	30	250	36 000	500	72 000
^{224}Ra	6	196	755	3500	1560	6900
^{232}Th	9	30	250	36 000	500	72 000
^{228}Ra	6	196	755	3500	1560	6900
^{238}U	1.24	38	89	158	175	400

Table 18: Range of tritium (as HTO) relative biological effectiveness (RBE) values for various endpoints (data extracted from Straume and Carsten, 1993)

Endpoint	RBE range	Number of studies
carcinogenesis	~1 to 1.8	6
genetic endpoints	~1 to 2.9	13
developmental and related effects	1.3 to 2.6	3
reproductive effects	1.4 to 2.9	5

Table 19: Realistic risk quotients (RQs) calculated for the potential toxicity of radionuclides released from uranium mines and mills to the environment

	Serpent River system		Cluff Lake area	Rabbit Lake area				Key Lake area	Beaverlodge Lake area	Maximum pre-operational mining areas [2]
	Maxima of area lakes	Quirke Lake[1]		Maxima of area lakes	Horseshoe Lake[1]	Collins Bay[1]	Wollaston Lake[1]			
fish	0.52		0.67	0.17				**1.2**	**5.6**	**1.1**
benthic invertebrates	**3**		**1.4**	**7.4**	**3.8**	0.17		0.59	**41**	0.20
algae	0.58	0.12	0.08	**13**			0.02	0.19	**6.1**	0.54
macrophytes	**1.18**	0.35	0.65	**26**			0.04	0.07	**12**	0.05
soil invertebrates			0.37					0.0004		0.91
litter invertebrates								0.12[3]		
								34[4]		
								3.4[5]		
small mammals (voles)								0.32[3]		0.34
								3.9[4]		
								2.2[5]		

	Serpent River system		Cluff Lake area	Rabbit Lake area				Key Lake area	Beaverlodge Lake area	Maximum pre-operational mining areas [2]
	Maxima of area lakes	Quirke Lake[1]		Maxima of area lakes	Horseshoe Lake[1]	Collins Bay[1]	Wollaston Lake[1]			
vegetation								0.13[3] **1.86**[4] 0.31[5]		
blueberries										0.90
Labrador tea										0.77
lichen								0.21		0.04

[1] Area lakes separated out to show spatial extent of potentially harmful effects on various taxa.

[2] Data from Cigar Lake, McClean Lake, Midwest Joint Venture and McArthur River mine areas before ore extraction and/or milling of ore commenced. The RQ of 1.1 is from the Midwest Joint Venture Project area.

[3] Site 1 control site.

[4] Site 2 southeast of above-ground tailings management facility.

[5] Site 3 south of Gerald Lake, receives yellowcake and ore dust from mining and milling facilities.

Table 20: Radiation dose and conservative risk quotients (RQs) for a mouse exposed to ^{222}Rn air emissions from uranium mines and tailings management facilities

Site	Date	^{222}Rn ($Bq \cdot m^{-3}$)	Dose[1] ($mGy \cdot a^{-1}$)	RQ[2]	Comment
Uranium mines					
Elliot Lake	1984–85	34.04	0.66	0.002	tailings site D11
		4.81	0.093	0.0002	background
Rabbit Lake	1989	185	3.58	0.009	EM3
	1991	48.1	0.93	0.002	EM5
Cluff Lake	1996	1716.9	33	0.083	mill grinding area
	1997	62.9	1.22	0.003	site 14 Cluff centre at high-volume air sampler
Key Lake	1985	11	0.21	0.001	
	1997	2	0.039	0.0001	
McClean Lake	1997	49.1	0.95	0.002	
Midwest Joint Venture Project	1989	270.1	5.225	0.013	
	1989	26 932.3	521.04	**1.3**	mine exhaust location
	1997	29.9	0.58	0.001	
Cigar Lake	1899	7.4	0.14	0.0004	background
Waste management area					
Welcome	1994	360	6.97	0.017	northeast gate

[1] A DCF of 0.000 264 was used to convert $Bq \cdot m^{-3}$ to $Gy \cdot a^{-1}$ using an RBE weighting factor of 40 for alpha radiation in calculating the radiation dose.
[2] RQs greater than 1 are in boldface type.

Table 21: Risk quotients (RQs)[1] calculated for radionuclide releases from uranium refineries and waste management facilities

	Refineries		**Waste management areas**					
	Blind River	Port Hope Harbour	Port Granby	Welcome	Chalk River Laboratories			
					Ottawa River	East Swamp Stream	Perch Lake Inlet 2	Duke Swamp[2]
mammals	–	–	–	–	–	–	–	**3.2**
fish	0.29[3]	1.0	0.41	0.31	0.003	**4.8**	0.17[3]	–
amphibians	–	–	–	–	–	–	–	0.5
benthic invertebrates	–	5.1	–	–	7×10^{-5} [4]	–	–	**4.6**
algae	0.11[3]	0.24	0.26	0.14	0.0004[3]	0.74	0.13[3]	–
macrophytes	0.27[3]	0.49	0.53	0.36	0.002[3]	0.72	0.09[3]	–

[1] RQs greater than 1 are in boldface type.
[2] Data from AECL (1999).
[3] Conservative RQs; other values are realistic RQs.
[4] Radiation dose based on radionuclide concentrations in beach sand.

Table 22: Conservative risk quotients (RQs), except for values marked with an R for realistic RQs, for potential toxicity of radionuclides released from nuclear generating stations (NGSs)

	Pickering NGS	Darlington NGS	Bruce NPD[1]	Gentilly NGS	Point Lepreau NGS
fish	0.0004 R	0.0001 R	0.001 R	0.003 R	0.19 R
benthic invertebrates	4×10^{-6} R	1.0×10^{-6} R	9.7×10^{-6}	0.0001 R	0.0001 R
algae	0.002	0.0002	0.0001	–	0.005
macrophytes	0.001	0.001	0.0003	0.001	0.001

[1] Nuclear Power Development.