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Part VI:

Guidance on Human Health Detailed Quantitative Radiological Risk Assessment ($DQRA_{RAD}$)

Federal Contaminated Site Risk Assessment in Canada

Canada

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FEDERAL CONTAMINATED SITE RISK ASSESSMENT IN CANADA

PART VI: GUIDANCE ON HUMAN HEALTH DETAILED QUANTITATIVE RADIOLOGICAL RISK ASSESSMENT ($DQRA_{RAD}$)

September 2010

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Contaminated Sites Division
Safe Environments Directorate

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PREFACE

The Federal Contaminated Sites Action Plan (FCSAP) is a program of the Government of Canada designed to ensure improved and continuing federal environmental stewardship as it relates to contaminated sites located on federally owned or operated properties. Guidance documents on human health risk assessment (HHRA) prepared by the Contaminated Sites Division of Health Canada, in support of the FCSAP, are available on our website and may also be obtained by contacting the Contaminated Sites Division at cs-sc@hc-sc.gc.ca.

This guidance document *(Federal Contaminated Site Risk Assessment in Canada, Part VI: Guidance on Human Health Detailed Quantitative Radiological Risk Assessment ($DQRA_{Rad}$)*) was prepared to provide guidance for custodial departments.

Ideally, federal sites contaminated with radioactive substances (radionuclides) would be remediated to an "essentially negligible" risk level. Health Canada's definition of "essentially negligible" is generally consistent with the Canadian Nuclear Safety Commission (CNSC) definition of "essentially negligible" (AECB, 1987). Similarly, the International Commission on Radiological Protection (ICRP) specifies a "minimum constraint" level; where annual doses are less than 0.01 mSv (10 μSv) the dose is acceptable, and it is not necessary to remediate the site further to reduce the dose. Radionuclides and radioactivity are naturally occurring, are ubiquitous in the natural environment, and the amounts vary from place to place.

Natural background radiation levels in Canada result in an average annual dose of about 2 mSv, but there is a wide variation from place to place. If a nominal risk coefficient for radiation risk of 5% per Sv or 5×10^{-5} per mSv is taken as applying at the magnitude of doses received from natural background radiation (i.e. a linear no-threshold model is used), this dose from natural background radiation could result in a lifetime risk of fatal cancer in the order of 2 mSv/year × 70 years × $5 \times 10^{-5} \approx 7.5 \times 10^{-3}$, or about 1/100. From 25% to 30% of Canadians die from cancer; thus, given the above assumptions, **natural** background radiation theoretically accounts for about 4% of background (fatal) cancers (i.e. 1/100 cancers attributable to natural background radiation compared with a rate of fatal cancer in the population, from all causes, of about 25/100).

For radioactivity, an annual dose limit of 1 mSv is used in establishing release limits for sources regulated by the CNSC. For closed-out/decommissioned facilities, an annual dose limit may be set at 1 mSv (for sites with some level of land use control), or at some fraction of 1 mSv for sites under unrestricted use. For example, Canadian Guidelines for Naturally Occurring Radioactive Materials (NORM), developed jointly by Health Canada and a Federal Provincial Territorial Committee, suggest an annual dose level of 0.3 mSv for unrestricted sites contaminated with naturally occurring radioactivity. The Canadian guidelines for drinking water quality for radionuclides employ an annual reference dose level of 0.1 mSv. The U.S. EPA has selected an annual level of 0.15 mSv and the U.S. NRC an annual level of 0.25 mSv as remediation goals for the unrestricted use of decommissioned sites (both of these dose levels are exclusive of the doses from radon). However, the U.S. EPA stipulates that risk characterization and remedial goals for radioactive substances at National Priority List (Superfund) *Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)* sites must target the 10^{-4}(which according to the U.S. EPA corresponds to a dose of 0.15 mSv) to 10^{-6} risk range for cumulative cancer risk. In particular, the EPA stipulates that**:**

> "Guidance issued by other organizations (e.g. NRC, DOE, ICRP, NCRP) may provide technical assistance; however the reader should exercise caution since some of these documents utilize a framework for risk management (e.g. allowable dose limits of 25, 100 or 500 mrem/yr) that EPA has determined is not suitable for use at CERCLA sites" (U.S. EPA, 1999, p.22).

Health Canada proposes a provisional "essentially negligible" annual dose level of 0.01 mSv above background for application at federal sites contaminated with radioactivity. The ICRP (2004) recommends this as a "minimum constraint" that should be considered for application in any situation. This dose rate is approximately equivalent to a risk of fatal cancer of 5×10^{-7} per year of exposure, and a risk for combined fatal cancers, non-fatal cancers, and serious hereditary effects of 7.3×10^{-7} per year of exposure. The corresponding lifetime (e.g. 70 years) risk of combined effects would be 5.1×10^{-5}. At federal contaminated sites with radionuclide contamination, an upper limit of 0.3 mSv above background should not be exceeded for unrestricted land use, per the NORM guidelines. Technical feasibility and socio-economic considerations are important considerations at all contaminated sites. For radionuclides, Health Canada stipulates that the ALARA principle (as low as reasonably achievable) should be enforced to ensure that human health risks are managed to a minimal level. Levels below 0.01 mSv are considered to be negligible and do not require further assessment.

This document provides guidance for human health risk assessment of radiological substances at federal contaminated sites. Specifically, this guidance provides a means of harmonization of chemical and radiological risk assessment and risk characterization of radiological doses at federal contaminated sites.

As is common with any national guidance, this document will not satisfy all of the requirements presented in every case by contaminated sites, custodial departments, or risk assessors. As the practice of HHRA advances, and as the FCSAP proceeds, new and updated information on various aspects of HHRA will be published. As a result, it is anticipated that revisions to this document will be necessary from time to time to reflect this new information. Health Canada should be consulted at the address below to confirm that the version of the document in your possession is the most recent edition and that the most recent assumptions, parameters, etc., are being used.

In addition, Health Canada requests that any questions, comments, criticisms, suggested additions, or revisions to this document be directed to: Contaminated Sites Division, Safe Environments Directorate, Health Canada, postal locator 4111A, 99 Metcalfe Street, 11th Floor, Ottawa, ON K1A 0K9. E-mail: cs-sc@hc-sc.gc.ca
See also: http://www.hc-sc.gc.ca/ewh-semt/contamsite/index_e.html.

ABBREVIATIONS AND ACRONYMS

ACRP	Advisory Committee on Radiation Protection
AECB	Atomic Energy Control Board
ALARA	as low as reasonably achievable
AMAD	activity median aerodynamic diameter
BIOMASS	Biosphere Modelling and Assessment Methods
BIOMOVS	Biospheric Model Validation Study
CCME	Canadian Council of Ministers of the Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CNSC	Canadian Nuclear Safety Commission
CSA	Canadian Standards Association
DCF	dose conversion factor
f_1	gut-to-blood transfer factor
FCSAP	Federal Contaminated Sites Action Plan
HHRA	human health risk assessment
HTO	tritium water vapour
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
NB	New Brunswick
NFLD/LAB	Newfoundland and Labrador
NORM	naturally occurring radioactive material
NS	Nova Scotia
PDF	probability distribution function
PEI	Prince Edward Island
PRG	preliminary risk-based guideline
RME	reasonably maximally exposed
RnD	radioactive decay products of radon
SAIC	Science Applications International Canada
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
U.S. EPA	United States Environmental Protection Agency
U.S. NCRP	United States National Council on Radiation Protection and Measurements
U.S. NRC	United States Nuclear Regulatory Commission
WL	working level
WLM	working level month

1.0 BACKGROUND AND CONTEXT

1.1 Background

Health Canada, the Canadian Council of Ministers of Environment (CCME), and provincial environmental and health regulatory agencies have common goals with respect to protecting human health from exposures to chemicals or radiation hazards found at contaminated sites. The common goals include the need for a coordinated and consistent approach to identify, assess, and manage health risks associated with contaminated sites. Guidance on site identification and assessment has been developed by CCME and includes:

- *National Classification System for Contaminated Sites* (1992)
- *Canadian Environmental Quality Guidelines* (2002)
- *Guidance Manual on Sampling, Analysis, and Data Management for Contaminated Sites* (1993)

Within the federal government, the Federal Contaminated Sites Action Plan (FCSAP) was established to assist in identifying, assessing and managing the risks at contaminated properties under the custodial care of Canadian federal government departments and agencies. A major emphasis of the FCSAP is to give priority for remediation or risk management to those sites and properties posing the greatest risks. Under the FCSAP, Health Canada was designated as an expert support department. In that role, Health Canada is required to provide detailed guidance on the assessment of human health risks posed by federal contaminated sites in Canada. The guidance presented herein related to sites contaminated with radioactivity is Part VI in a series of documents published by Health Canada on federally contaminated site risk assessment.

This document was prepared in support of the FCSAP – a program designed to ensure improved and continuing federal environmental stewardship as it relates to contaminated sites located on federally owned or operated properties. As is common with national guidance, this document will not satisfy, nor is it intended to satisfy, all of the requirements presented by contaminated sites or risk assessors in every case.

As the identification and assessment process of contaminated sites has matured, the use and application of human health risk assessments (HHRAs) has increased. Currently, risk assessments are conducted following guidance published by Health Canada, as well as several provinces. The need for a coordinated and consistent national approach has led to the development of this guidance document.

1.2 Regulatory Context

Most provinces have developed regulations, policies and/or procedures for assessing and managing contaminated sites. A review of provincial regulations and policies relating to contaminated sites was undertaken on behalf of Environment Canada by Science Applications International Canada (SAIC, 2005). Currently, provinces only address contamination with inorganic elements and organic chemicals. The various provincial and territorial regulatory frameworks include:

British Columbia – *Environmental Management Act, Contaminated Sites Regulation.*

Alberta – *Environmental Protection and Enhancement Act*; *Guidelines for the Designation of Contaminated Sites*

Manitoba – *Contaminated Sites Remediation Act*; *Contaminated Sites Remediation Regulations*; *Guidelines for the Designation of Contaminated Sites in Manitoba*; *Guideline for Site Environmental Investigations in Manitoba*

Saskatchewan – *Environmental Management and Protection Act, 2002*

Ontario – *Environmental Protection Act*; *Ontario Regulation 153/04*

Quebec – *Environmental Quality Act*; *Land Protection and Rehabilitation Regulation*; *Spill Contingency Planning and Reporting Regulations*; *Soil Protection and Contaminated Sites Rehabilitation Policy*

New Brunswick–- *Clean Environment Act*; *Petroleum Product Storage and Handling Regulation*; *Water Quality Regulation*; *Guideline for the Management of Contaminated Sites (version 2.0)*; *Record of Site Condition (version 2.0)*

Nova Scotia – *Environment Act*; *Guidelines for Management of Contaminated Sites in Nova Scotia*; *Guidelines for the Designation of Contaminated Sites*

Prince Edward Island – *Environmental Protection Act*; *PEI Petroleum Contaminated Sites Remediation Guidelines*

Newfoundland and Labrador – *Environmental Protection Act*; *The Storage and Handling of Gasoline and Associated Products Regulations*; *Heating Oil Storage Tank System Regulations*; *Contaminated Sites Clean-Up Criteria*

Yukon – *Environment Act*; *Contaminated Sites Regulation*; *Spills Regulations*; *Special Waste Regulations.*

Northwest Territories – *Environmental Protection Act*; *Spill Contingency Planning and Reporting Regulations*; *Environmental Guideline for Contaminated Site Remediation, 2003.*

Nunavut –- *Environmental Protection Act*; *Guideline for Spill Contingency Planning (draft), 2004*; *Environmental Guideline for Contaminated Site Remediation, 2002.*

Contaminated sites that are the property of, or in the custodial care of, federal government departments are not subject to provincial regulation. For federal sites, the Treasury Board Secretariat of Canada has established a series of policies regarding assessment and remediation/risk management (http://www.tbs-sct.gc.ca/pubs_pol/dcgpubs/realproperty/fcsmp-gscf_e.asp) and (http://www.tbs-sct.gc.ca/pubs_pol/dcgpubs/realproperty/tbfcsswlip_e.asp). The FCSAP is not a regulatory initiative, but rather a Treasury Board-funded and -directed program to improve federal environmental stewardship with respect to legacy sites. More details on the FCSAP are available at http://www.federalcontaminatedsites.gc.ca/.

In Canada, the Canadian Nuclear Safety Commission (CNSC), which succeeded the Atomic Energy Control Board (AECB) in May 2000, regulates the use of nuclear energy and materials, as well as nuclear wastes, to protect health, safety, security, and the environment. In addition, Health Canada through the Federal Provincial Territorial Radiation Protection Committee is responsible for the management of naturally occurring radioactive materials (NORM). The assessment and remediation of inactive federal sites contaminated with radioactivity, on the other hand, has generally "fallen between the cracks." However, the advent of the FCSAP in 2003 established a mechanism to fund the assessment and remediation of legacy federal sites contaminated with radioactivity. The federal government has liability for a number of sites across Canada that are contaminated with radioactivity, such as the site of a former uranium mine at Port Radium on Great Bear Lake in the Northwest Territories.

1.3 Purpose and Audience

The primary users of this risk assessment guidance manual are anticipated to be risk assessors who conduct quantitative HHRAs at federally contaminated sites in Canada under the FCSAP program. The manual will also be useful to risk assessment reviewers (regulators), risk managers (regulators or site owners), and other professionals in industry who are involved in the site investigation and remediation process. It is important to provide guidance to risk assessors who conduct quantitative HHRAs at contaminated sites containing radioactive material under the FCSAP program to ensure consistency, comparability, and defensibility of those assessments.

Individuals with various scientific backgrounds and experience in radiological risk assessments will look to this manual for guidance on how to conduct a radiological risk assessment. Therefore, this manual is designed not only as a "how to" document for conducting typical radiological risk assessments, but also as an educational document explaining "why" certain things are done or steps are taken, etc. The manual also provides a background to the principles applied to radiological risk assessments and a comparison with conventional chemical risk assessments. A brief discussion of more sophisticated methodologies for conducting radiological risk assessments is also provided. The manual provides considerable background information relevant to radiological dose and risk assessment, and outlines the various steps in the radiological risk assessment process. Examples are provided to illustrate these steps. Some case studies are also provided (see Appendix C).

The user of this manual will learn how to:

- apply the radiological risk assessment framework to "simple" sites, including documenting the assumptions, rationale, decisions, and processes to allow the risk assessment reviewer to track, re-trace, and reproduce the path followed in the application of this risk assessment guidance;
- determine whether the data collected for the site are appropriate and of sufficient quality and quantity to be used in the assessment;
- decide when to use a **deterministic** or a **probabilistic** approach in the risk assessment;
- select appropriate values for input parameters and necessary assumptions, or probability distribution functions (PDFs), in the case of a probabilistic assessment that consider site-specific conditions and land use plans;
- identify appropriate references, databases, modelling methods, and other resources that may be required in the process;
- select appropriate default dose-conversion factors to be used in the assessment;
- select appropriate dose limits to be used in the assessment; and
- provide the custodial federal department for the site with a clear understanding of the potential risk(s) to assist in communication with risk managers, regulators, and other stakeholders, and to identify appropriate risk management measures (including remediation).

It is necessary that the risk assessor and risk assessment reviewer have the scientific training and background to follow the proposed framework and to apply appropriate judgment and assumptions as required. The manual is not intended to instruct those unfamiliar with the various scientific concepts and areas of knowledge necessary for the completion of a risk assessment. Those areas where professional judgment is essential include:

- determining the relevance of existing data and, if appropriate, designing a program to collect additional data;

- formulating and designing the risk assessment;
- assessing communication requirements;
- evaluating and selecting appropriate models and analytical methods;
- developing appropriate input parameter assumptions;
- evaluating uncertainty and assessing the value of collecting additional information; and
- interpreting the significance of estimated risks.

Health Canada has published, or will be publishing, guidance on some of these elements and aspects of risk assessment. However, it is still essential that a trained and experienced risk assessment practitioner conduct the work.

1.4 Differences Between Chemical and Radiological Risk Assessment

Ionizing radiation (radiation) and genotoxic chemicals (chemicals) can both cause cancer. The methods of assessing risks from radiation and chemicals are well developed, and generally similar in principle. In order to have a risk from radiation or chemicals, there must be a source, receptor (human), and one or more pathways from the source to the receptor to effect exposure. Human exposure to radiation or chemicals is evaluated with the help of an exposure pathways analysis that quantifies potential exposure by evaluating (with the help of data and/or models) the concentrations of radionuclides or chemicals in air, water, soil, food or other media; and the fate and mobility of elements in the environment. It also takes account of the characteristics of the exposed people (receptors).

One of the important features of radionuclides is that they undergo radioactive decay with the emission of energy and radioactive particles, changing at the same time into other radioactive species, each with its own unique physical and chemical characteristics.

Radionuclides and chemicals can be taken into the body via inhalation, ingestion, or dermal absorption. However, unlike chemicals, it is not necessary that the radionuclide be taken into the body to elicit an effect; this is because exposure can occur from sources external to the body that release radiation (primarily gamma radiation) that can penetrate the skin and deposit energy on body tissues and organs. An example of this would be the gamma radiation emitted from sites contaminated with radium-226 (Ra-226), a member of the uranium-238 (U-238) decay chain.

For both radioactivity and genotoxic carcinogenic chemicals, risk assessors assume, in the absence of evidence to the contrary, a linear, no-threshold relation between dose and effects. In other words, there is assumed to be some risk at any level of exposure other than zero. For both radiation and chemicals, considerable uncertainty is associated with the dose-effect relationship. For radiation, the risks are largely derived from epidemiological studies of people exposed to high levels of radiation in the past; examples include the Japanese atomic bomb survivors, radium dial painters, uranium miners, and various exposure groups arising from past medical practices (e.g. see the discussions of human epidemiology in UNSCEAR, 2000). The risks from radiation depend upon the type of radiation and the sensitivity of each organ exposed to the radiation. Fortunately for radiation risk assessments, it is possible not only to estimate the cancer risk for an individual organ, but also to estimate an aggregate risk that takes into account the risk to all organs from all radionuclides and routes of exposure. In such cases, the radiation exposures are referred to as **effective doses** in which the relative sensitivities of the various organs are taken into account with the use of organ weighting factors. In addition to having well-established risk estimates for fatal cancer, the International Commission on Radiological Protection (ICRP) has also recommended risk coefficients for cancer incidence and serious hereditary effects (ICRP, 1991).

For carcinogenic chemicals, dose-effect relationships are often extrapolated from animals exposed to relatively high doses, typically greater than doses associated with contaminated site exposures. Extrapolation of cancer risk from animals to humans is generally based on the upper 95% confidence limit of the curve fit to experimental data. The use of the upper confidence limit in such extrapolation reflects an application of the precautionary principle, and helps to ensure, to the degree possible, that the actual risk is unlikely to be greater than that predicted and most likely is lower than that predicted. Unlike ionizing radiation, it is not possible at this time to develop risk assessments for combined exposures to multiple chemicals affecting multiple organs other than to simply add the estimated risks for chemicals affecting like organs, causing like cancers, and/or acting by similar modes of action.

In 1995, a Joint Working Group from the former AECB Advisory Committee on Radiation Protection (ACRP), AECB staff, Health Canada staff, and staff from the Ontario Ministry of the Environment carried out a comprehensive evaluation of the similarities and differences in assessing and managing cancer risks from radiation and chemical hazards. Although the general approaches for risk assessment of ionizing radiation and cancer-causing chemicals are similar, differences still exist. The Joint Working Group considered that risk assessment and risk management methods for both radiation and chemicals were well developed, and that current risk management strategies for both radiation and chemicals provided a high degree of health protection. It also noted that it was not possible at that time (1995) to determine if radioactivity or carcinogenic chemicals present the greatest risk of cancer (HC/AECB, 1998). Although this is still largely true, steps have been taken in this guidance to harmonize the approach to both chemical and radiological risk assessments. For example, as will be discussed later, this manual provides a harmonization of receptor

characteristics used in the two assessments. Greater harmonization in the level of risk considered to be "essentially negligible" is also introduced.

1.4.1 Dose versus risk

The risk of developing cancer from exposure to radiation depends on the type of radiation and the sensitivity of the specific organ that is exposed to the radiation. Therefore, radiation exposures are generally calculated as an **effective dose** that takes into account the type of radiation and radiation sensitivity of the organ by using weighting factors developed by the ICRP (e.g. ICRP, 1991). In effect, the system of dose/risk estimation for radiation allows for the "risk normalized" addition over all exposure pathways, radionuclides, and organs. The use of these weighting factors also allows for weighting of different cancers; for example, a fatal cancer has a larger weight than a cancer that is termed "curable" (non-fatal).

The ICRP Publication 60 (ICRP, 1991) recommended a risk coefficient (cancer incidence rate) of 5×10^{-5} per mSv for fatal cancers (average over all age groups) and a risk coefficient of 1×10^{-5} per mSv for non-fatal cancers. These risk coefficients represent the lifetime risks from exposure. For example, for 1 year of exposure to 1 mSv, the lifetime risk of fatal cancer would be in the order of 5×10^{-5} and for 70 years of exposure[1] about 3.5×10^{-3}. The ICRP also recommended a lifetime (detriment) risk factor of 1.3×10^{-5} per mSv for serious hereditary effects. As a result, the total lifetime (detriment) risk is 7.3×10^{-5} per mSv (e.g. $5 \times 10^{-5} + 1 \times 10^{-5} + 1.3 \times 10^{-5}$). Thus, the effective dose provides the basis for estimating lifetime total risk from radiation exposure from any type of radiation, any distribution of dose, or any route of exposure be it from internal or external radiation. This dose is then generally compared with regulatory dose limits, such as those provided by the CNSC, to determine if an individual is at "unacceptable" risk from exposure to radiation/radioactive materials. In practice, the risk coefficient of 5×10^{-5} per mSv for lifetime risk of fatal cancer is typically used in such analyses. In Canada, the CNSC has established an annual dose limit of 1 mSv **above background**, and **exclusive of medical radiation** as the public dose limit for its licensees (CNSC, 2000). In considering the dose limit, it is important to remember that the combined exposure from all radionuclides and all pathways be considered.

For carcinogenic chemicals, guidelines for maximum acceptable concentrations are often defined for lifetime risk of cancer incidence in the order of 1 in 100,000 (1×10^{-5}) to 1 in 1,000,000 (1×10^{-6}), and guideline values associated with higher risks would only arise from contaminants that occur in nature (e.g. arsenic, which has a new proposed drinking water guideline of 5 µg/L). This concentration is associated with a lifetime risk of developing internal cancers of about 1 in 10,000 (HC, 2004a). It should also be emphasized that the application of the chemical risk guidelines is most often for **individual** cancer-causing chemicals whereas, for ionizing radiation, the risk from all radionuclides and exposure pathways is considered in a **single combined measure**. That being said, most federal sites contaminated with known carcinogens usually contain only one or a few carcinogenic substances. A recent analysis of federal sites conducted for Health Canada (Franz Environmental Inc., 2005) showed that arsenic and benzene and benzo(a)pyrene occur together in only 0.2% of federal sites examined, benzo(a)pyrene and arsenic occur together in 2.3% of sites examined, and benzo(a)pyrene and benzene occur together in 0.7% of sites.

1.4.2 International context

There is greater international standardization of radiation risk assessment than for chemical risk assessment. A great deal of work has been carried out over the past 50 years in developing methods, data, and models for assessing exposure to radioactivity and radiation, and for estimating risk to people from ionizing radiation. The United Nations Scientific Committee on the Sources and Effects of Ionizing Radiation (UNSCEAR) has the United Nations mandate to assess and report on levels and effects of exposure to ionizing radiation. UNSCEAR systematically reviews worldwide information and publishes detailed reports that form the basis for scientific evaluations by various international and national organizations. Annex A, Dose Assessment Methodologies, of the 2000 UNSCEAR report reviewed the worldwide procedures used by UNSCEAR to estimate doses from radionuclides in the environment. The annex models are noted to provide the basis for "reasonably accurate estimates of dose in many applications." However, UNSCEAR recommends that "site-specific data should be used as appropriate and when available."

[1] For ionizing radiation, the risk (detriment in ICRP terms) for a member of the public is based on a 70-year integration of dose following the intake of a radionuclide. For example, if a person were to ingest "X" Bq Ra-226 in a given year, a portion of the Ra-226 would be eliminated from the body and a portion would be distributed among body tissues in accordance with well-established metabolic/biokinetic models. In addition, a portion would undergo radioactive decay. The portion of the Ra-226 remaining in the body (and radioactive decay products) will continue to irradiate body tissues. The ICRP dosimetric models estimate the cumulative dose to body organs/tissues for 70 years following the year of intake. By convention, all of the dose estimated from the 70-year integration is assigned to the year of intake. This methodology is different from the use of the 75-year lifetime exposure used in chemical risk assessments; however, the use of the 70-year lifetime in radiological assessments is similar for practical purposes.

In addition to the description of models and data provided by UNSCEAR, the published literature in this field is extensive, and numerous authors have published reports on methods, models, and data. For example, a 1987 Canadian standard provides both guidance and a methodology for estimating radiation doses from radionuclides released to the air or water from Canadian deuterium–uranium reactor nuclear power plants (CSA, 1987). Much of the information concerning exposure pathways, models, concepts, and methods in this document remain valid and relevant today. In addition, much information on exposure pathways and risk assessment methods and data are also available from the United States, including for example, the United States Nuclear Regulatory Commission (U.S. NRC) Mildos Code (Strenge and Bander, 1981) for assessing exposure from uranium mill tailings. The U.S. NRC has published more recent guidance specific to the evaluation of radioactive contaminated sites (U.S. NRC, 1992). Similarly, the United States National Council on Radiological Protection (U.S. NCRP) has published extensively on the methods and data for exposure pathways analysis, dose assessment, and uncertainty analysis (U.S. NCRP, 1996). Other groups, notably the International Atomic Energy Agency (IAEA) have also published models and data for exposure pathways analysis and dose assessment (e.g. IAEA, 1982, 1994) and supported international comparisons of environmental data and dose models, including those of the Biospheric Model Validation Study (BIOMOVS, 1996a, 1996b) and the Biosphere Modelling and Assessment Methods (BIOMASS) Programme, 2001) in which Canada has played a prominent role.

1.4.3 Canadian context

As discussed previously, the CNSC and Health Canada through the Federal Provincial Territorial Radiation Protection Committee have oversight over radiological issues in Canada. However, the CCME Soil Quality Guidelines Task Group is mandated to develop Canadian national soil quality guidelines and risk assessment guidance for contaminated sites, including those contaminated with radioactivity. For facilities that require CNSC licences, the CNSC dose limits prevail. These agencies have a general consensus as to the framework and methods that are routinely used in radiological risk assessments; the methods are discussed later in this manual. Federal contaminated sites represent a unique regulatory conundrum in that they are not subject to provincial regulation, and no formal federal regulatory framework addresses them. Hence, the Treasury Board has initiated the FCSAP as a means of rectifying these federal environmental liabilities.

1.5 Organization of Document

This manual is not only a “how to” guide, but it also provides extensive background information on radiation, as well as issues related to radiological risk assessments. Suggested values and methodologies to be used or considered by the risk assessor are provided in the following sections of the report:

Section 1.0 Background and Context
Provides an overview of radiological risk assessments and what will be presented in this manual.

Section 2.0 Overview of Radiological Risk Assessment Framework
Highlights the concepts of radiological risk assessment, including discussions on radiation, radiation protection, steps in the assessment, and a comparison with traditional chemical risk assessments.

Section 3.0 Problem Formulation
Discusses the problem formulation stage of the radiological risk assessment and all the information that needs to be considered before embarking on the radiological risk assessment.

Section 4.0 Fate and Transport Modelling
Discusses the Exposure Assessment steps, including fate and transport modelling, and the determination of concentrations in various environmental media. Both deterministic and probabilistic methods are discussed.

Section 5.0 Radiation Dose Assessment
Discusses the various factors used in the dose calculations of a radiological risk assessment.

Section 6.0 Dose Characterization
Provides information relating to dose limits, and how they should be considered and applied in radiological risk assessments.

Section 7.0 Risk Characterization and Harmonization
Discusses the conversion of annual and lifetime dose to annual and lifetime risk levels, and the current steps being taken to harmonize chemical and radiological risk.

Section 8.0 Addressing Uncertainty
Discusses the uncertainties associated with the various steps of the risk assessment.

Appendices Provide support discussion and information.

Appendix A. Generic Factors to Be Considered in Risk Assessment

Appendix B. Recommended Default Receptor Characteristics for Use in Radiological/Dose Risk Assessment

Appendix C. Example – Radiological Risk Assessment for Hypothetical Mine Site

Appendix D. References on Consumption of Fish and Wildlife by Aboriginal People of Northern Canada

2.0 OVERVIEW OF RADIOLOGICAL RISK ASSESSMENT FRAMEWORK

2.1 Introduction and Radiation Preliminaries

The focus of this document is HHRA at federal sites contaminated with radioactive substances. Before discussing the approach to HHRA for radioactive contaminated sites, it is useful to comment on certain aspects of radioactive substances and the radiation that distinguishes them from chemicals.

2.1.1 Commonly used terms

Some terms commonly seen in radiological risk assessments are presented in Table 2.1. For further information, the reader is referred to published reports such as those of the ICRP (e.g. ICRP, 1991), the IAEA (e.g. IAEA, 1982), the U.S. NCRP (e.g. U.S. NCRP, 1984, 1989, 1993), and UNSCEAR (e.g. UNSCEAR, 2000).

Table 2.1 Commonly Used Terms

Absorbed dose (*D*)	The mean energy imparted by ionizing radiation to matter per unit mass. The special SI unit of absorbed dose is the gray (Gy); the conventional unit is the rad (1 rad = 0.01 Gy).
Activity measured in becquerel (Bq)	One nuclear disintegration per second; the name for the SI unit of activity (1 Bq = 2.7×10^{-11} Ci)
Curie (Ci)	3.7×10^{10} nuclear disintegrations per second; the name for the conventional unit of activity (1 Ci = 3.7×10^{10}Bq)
Decay product(s)	A radionuclide or a series of radionuclides formed by the nuclear transformation of another radionuclide; in this context, referred to as the parent
Dose conversion factor (DCF)	The dose equivalent per unit intake of radionuclide
Dose equivalent (*H*)	The product of the absorbed dose (*D*), the quality factor (*Q*), and any other modifying factors (N). The SI unit of dose equivalent is the sievert (Sv); the conventional unit is the rem (1 rem = 0.01 Sv)
Effective dose	The sum over specified tissues of the products of the dose equivalent in a tissue or organ (T) and the weighting factor for that tissue
External radiation	Radiation incident upon the body from an external source
Gray (Gy)	The SI unit of absorbed dose (1 Gy = 1 Joule kg^{-1} = 100 rad)
Half-life (physical, biological, or effective)	The time for a quantity of radionuclide (i.e. its activity) to diminish by a factor of a half (because of nuclear decay events, biological elimination of the material, or both)
Internal radiation	Radiation emitted from radionuclides distributed within the body
Ionizing radiation	Any radiation capable of displacing electrons from atoms or molecules, thereby producing ions
Nuclear transformation	The spontaneous transformation of one radionuclide into a different nuclide or into a different energy state of the same nuclide
Quality factor (Q)	The principal modifying factor that is employed in deriving dose equivalent, H, from absorbed dose, *D*; chosen to account for the relative biological effectiveness (RBE) of the radiation in question, but to be independent of the tissue or organ under consideration, and of the biological endpoint. For radiation protection purposes, *Q* is determined by the linear energy transfer (LET) of the radiation.
Radioactive equilibrium	Where such radionuclides in a radioactive decay chain is present at the same activity
Weighting factor (W_T)	Factor indicating the relative risk of cancer induction or hereditary defects from irradiation of a given tissue or organ; used in calculation of effective dose equivalent and committed effective dose equivalent
Sievert (Sv)	The SI unit of effective dose

2.1.2 Natural background radiation and radioactivity

Regardless of where people live or work, they are exposed to natural sources of radiation. The magnitudes of these sources vary greatly both in time and space, and are mainly attributable to ionizing radiation from cosmic rays; naturally occurring radionuclides in air, water, and food; and gamma radiation from radioactive material in the soil, rocks, and building materials used in homes. (see Figure 2.1).

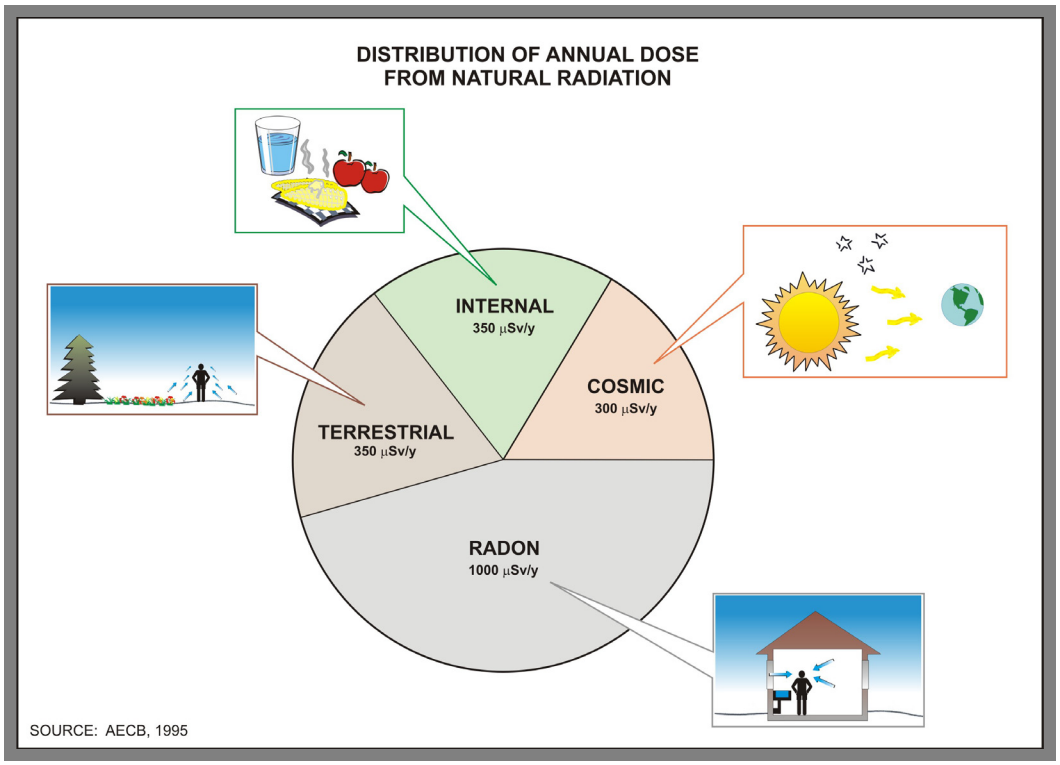
The Earth is continually bombarded by cosmic rays or high-energy particles that originate in outer space. These cosmic rays interact with atmospheric constituents, producing a cascade of interactions contributing to cosmic ray exposures that decrease in intensity with depth in the atmosphere, from aircraft altitudes to ground level. People at high latitudes and high altitudes receive the highest cosmic radiation doses owing to the focusing of cosmic rays to the Earth's poles by its magnetic field and the shielding (or lack of it) provided by the atmosphere. In Canada, the average annual dose from cosmic radiation is approximately 300 μSv.

Uranium and thorium series radionuclides and potassium-40 (K-40) are naturally occurring radionuclides found everywhere in soil, rocks, and sediments. They transfer naturally into vegetation and terrestrial and aquatic animals, and transfer to humans in food, water, and air. In Canada, the average annual dose from internal naturally occurring uranium and thorium series radionuclides and K-40 is approximately 350 μSv.

The naturally occurring radionuclides present in soils, rocks, and building materials used in homes also contribute to the external gamma radiation to which all people are exposed. In Canada, the average annual dose from gamma radiation from indoor and outdoor sources is approximately 350 μSv.

Radon gas and its radioactive decay products contribute the highest annual dose from naturally occurring radioactivity, estimated at about 1,000 μSv on average in Canada. Radon gas is a product of the decay of uranium series radionuclides in soil. Radon gas passes through foundation walls into building basements and accumulates to higher levels indoors than outdoors. The average annual dose from radon and its radioactive decay products in the air of houses in different Canadian cities ranges from approximately 200 to 2,200 μSv, depending on the concentration of radionuclides in soil, rock, and groundwater, as well as building ventilation rates.

Figure 2.1 Distribution of Annual Dose from Natural Radiation



Radioactivity and Radiation

- Radioactivity and radiation are everywhere.
- Every person is exposed to natural background radiation at levels of about 2 mSv/year.
- About half of the dose from natural background radiation is from radon.

One important aspect of radioactivity and radiation is that they are ubiquitous; this is different from many chemicals with the exception of arsenic and other naturally occurring mineral elements and certain other chemicals.

Grasty and LaMarre (2004) have reviewed published information on effective doses to Canadians from natural sources of radiation and report the effective doses for Canada, Toronto, and Winnipeg as 1769 µSv (1.8 mSv), 1554 µSv (1.6 mSv), and 4022 µSv (4.0 mSv), respectively. It is possible that much of the variability is associated with inhalation doses from radon-222 (Rn-222) and its short-lived decay products.

2.1.3 Radioactive decay chains

As indicated earlier, through the process of radioactive decay, one radioactive element is transformed into another, producing a "decay product" or a series of decay products with different physical and chemical characteristics from the parent in the radioactive decay series. The decay product(s) may also be radioactive. For example, the radioactive isotope Ra-226 decays into a (noble) radioactive gas, Rn-222. Figure 2.2 illustrates the concept by providing an illustration of the decay chain from the naturally occurring U-238 decay chain. In nature, it is most common that the U-238 decay chain is in equilibrium; this means that all of the (solid) radionuclides are present at the same concentration (e.g. for 1 Bq/g U-238, there will be 1 Bq/g U-234, 1 Bq/g Th-230, 1 Bq/g Ra-226, etc.). Figure 2.3 shows the U-235 decay series, and Figure 2.4 shows the thorium-232 (Th-232) decay series, both of which also occur in nature.

Figure 2.2 Uranium-238 Decay Series

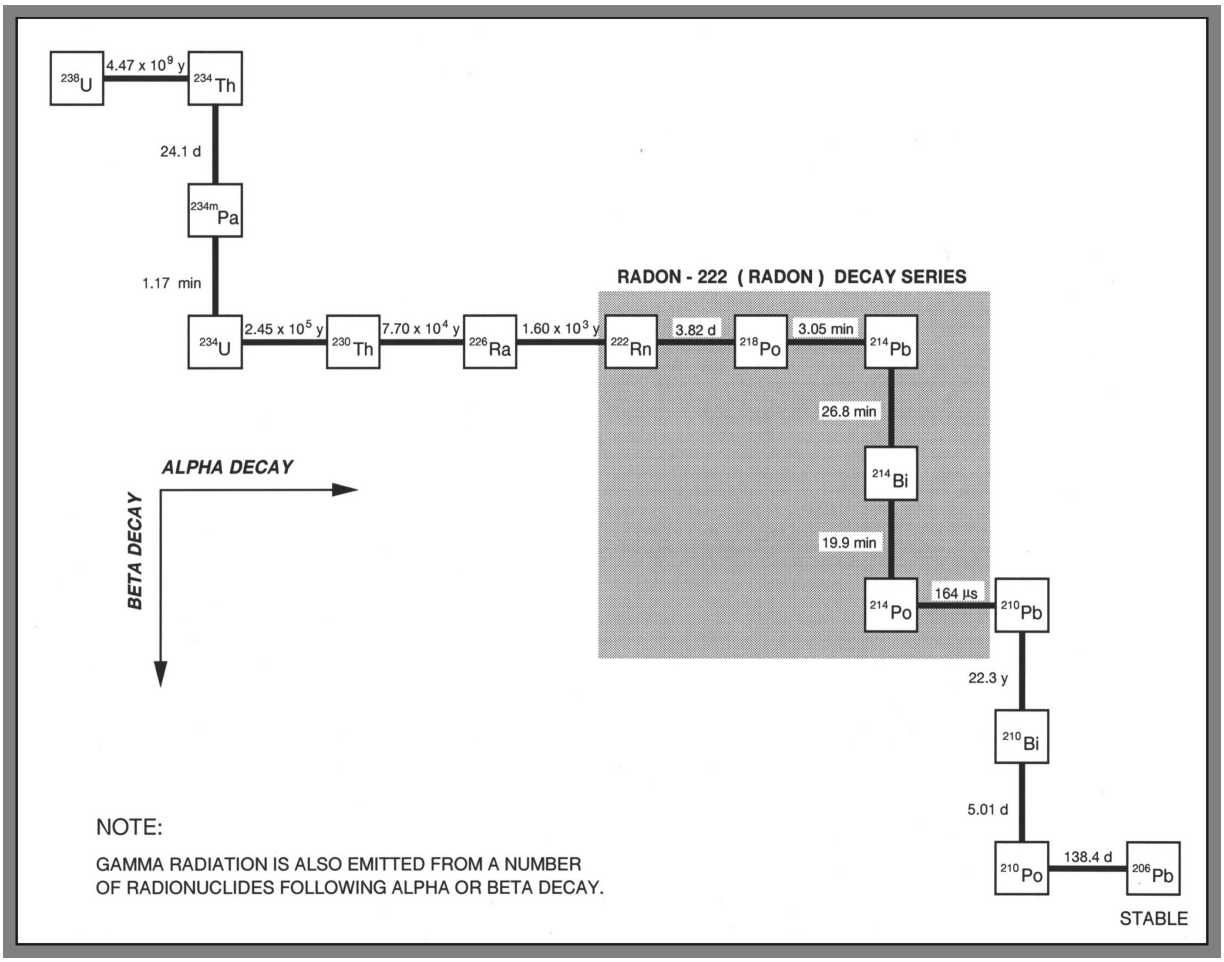


Figure 2.3 Uranium-235 Decay Series

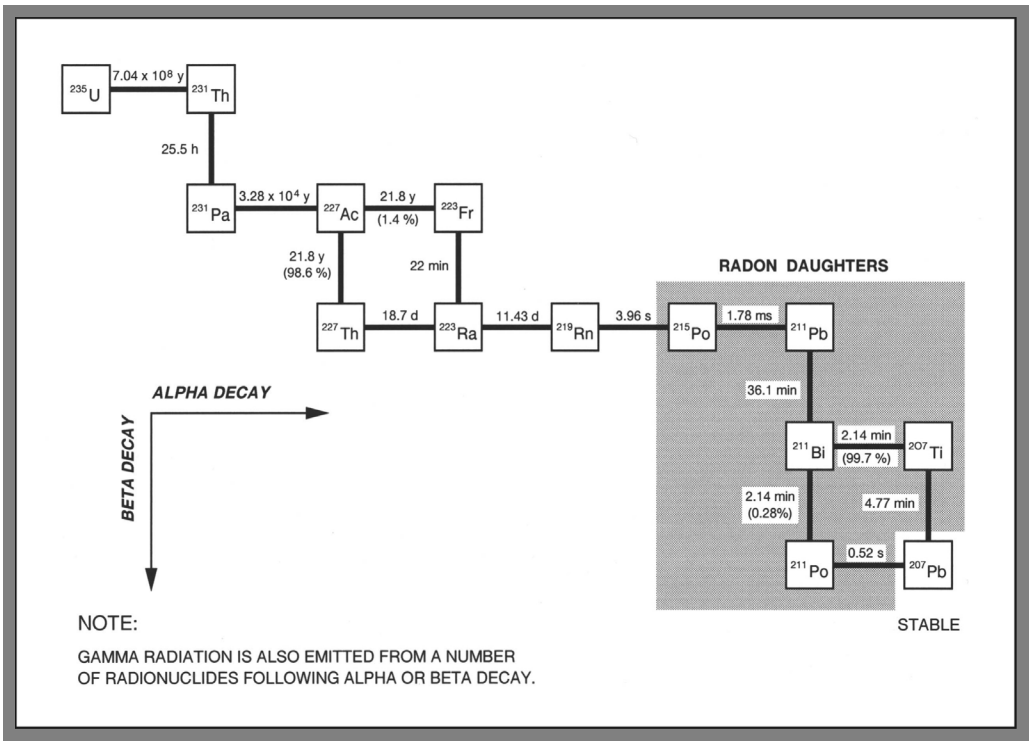
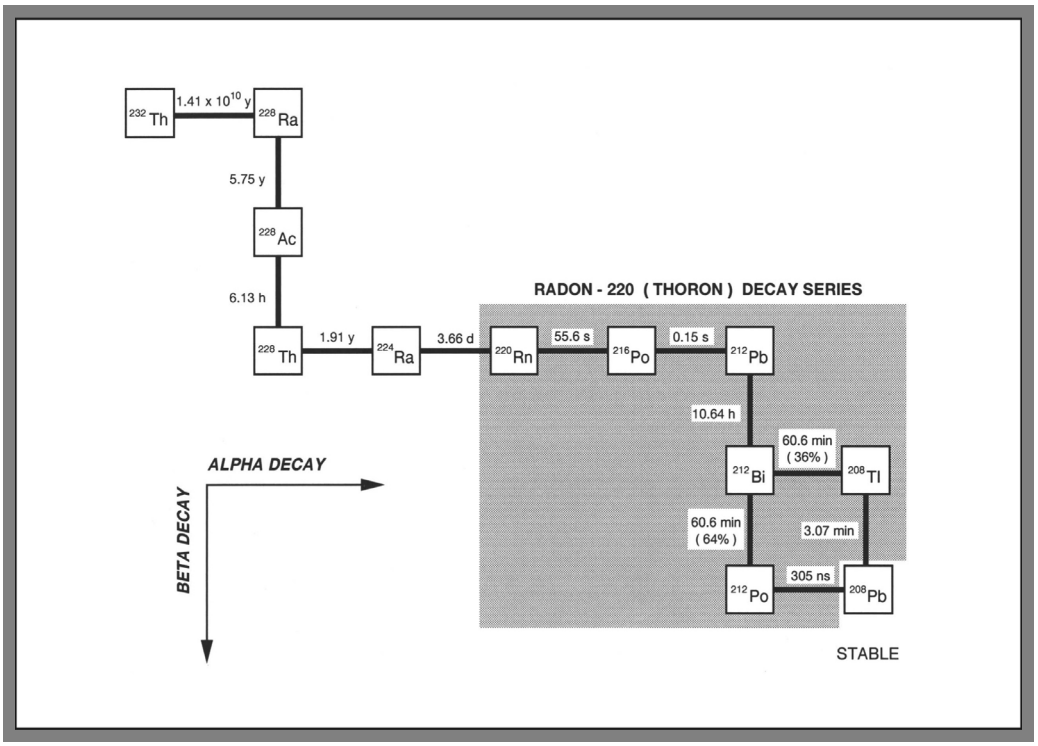


Figure 2.4 Thorium-232 Decay Series



Radioactive Decay

- Radionuclides undergo radioactive decay, emit radiation and/or radioactive particles and are transformed to a decay product with different physical, chemical, and radiological characteristics.
- Radioactive decay chains where all radionuclides in the chain are present at the same activity (Bq/g) are said to be in equilibrium.
- In assessing risks involving radionuclide decay chains, such as uranium-238, it is important to understand whether or not equilibrium exists.

In some situations, the (radioactive) equilibrium has been altered by nature or human; therefore, not all of the decay series radionuclides are present at the same activity. For example, in uranium mining, the objective is to remove uranium from the ore and send it for further processing. In this case, close to 90% to 95% of the uranium might have been removed in the uranium mill, with the remaining 5% discharged to the tailings along with all of the Th-230, Ra-226, etc. The radiological characteristics of the tailings would be in the proportions of 0.05 Bq/g U-238, 0.05 Bq/g U-234, 1 Bq/g Th-230, 1 Bq/g Ra-226, etc., with the actual numeric values depending on the grade of the ore. Note that all uranium isotopes behave chemically in the same fashion, all thorium isotopes behave chemically in the same fashion, all lead isotopes behave chemically in the same fashion, etc.

Each gram of natural uranium contains 1.295×10^4 Bq/g each of U-238 and U-234, and 5.69×10^2 Bq/g U-235, for a total specific activity of 2.65×10^4 Bq/g (i.e. $2 \times 1.295 \times 10^4 + 5.69 \times 10^2$). U-238 represents almost all (more than 99%) of the weight of natural uranium.

In addition to the two uranium decay chains (i.e. U-238 and U-235), Th-232 is also the head of a radioactive decay chain (see Figure 2.4) with a specific activity of about 4.06×10^3 Bq/g.

Other radionuclides, such as potassium-40 (K-40), tritium (as tritium water vapour [HTO]), and carbon-14 (C-14), occur in nature; the characteristics of such radionuclides will be discussed as needed in later sections.

In evaluating an exposure scenario, because the doses per unit intake depend on the radionuclide and on the route of exposure (among other factors), it is important to assess whether or not the radionuclides in a decay chain (such as the U-238 decay chain) are in equilibrium and which of the decay chain radionuclides may need to be considered in a particular risk assessment. In the evaluation of uranium mining activities, as at Port Radium, Northwest Territories, for example, it is common to consider natural uranium, Th-230, Ra-226, lead-210 (Pb-210), and polonium-210 (Po-210).

2.1.4 Types of radiation and radioactivity

In undergoing radioactive decay, various types of radiation are emitted. Each type of radiation has different physical characteristics and differs in its ability to cause biological damage. At sites that are being considered under the FCSAP program, the most likely forms of radiation and radioactivity to be encountered are as follows:

Alpha particles – These are doubly charged cations, composed of two protons and two neutrons (i.e. helium nuclei), that are ejected mono-energetically from the nucleus of an atom. Because of their relatively large mass and charge, alpha particles tend to ionize nearby atoms quite readily, expending their energy in short distances. Alpha particles will usually not penetrate an ordinary sheet of paper or the outer layer of skin. Thus, alpha particles only represent a hazard when taken into the body where their energy is completely absorbed by small volumes of tissues.

Beta particles – These are electrons ejected at high speeds from the nucleus of an atom undergoing radioactive decay to a proton and an electron. Unlike alpha particles, beta particles are not emitted with discrete energies, but are ejected from the nucleus over a continuous energy spectrum. Beta particles have a larger range than alpha particles, and unshielded beta sources can constitute external hazards if the beta radiation is within a few centimetres of exposed skin surfaces and if the beta energy is greater than 70 keV. Internally, beta particles have a much greater range than alpha particles in tissue. However, because they cause fewer ionizations per unit path length, beta particles deposit much less energy to small volumes of tissue, and consequently inflict less damage than alpha particles.

Gamma radiation – This radiation consists of photons emitted from the nucleus of an atom undergoing radioactive decay. (X-rays, which are extra-nuclear in origin, are identical in form to gamma rays, but have slightly lower energy ranges.) When gamma radiation passes through a human tissue or organ, a portion of the energy of the gamma radiation is absorbed into the tissue. The amount of energy absorbed by the tissue depends on the characteristics of both the tissue and the gamma radiation. As discussed below, it is the dose (the amount of energy) absorbed by a tissue that is relevant to risk assessments. Because of its large range and penetration ability, gamma radiation is of most concern as an external hazard.

2.1.5 Biological harm and dose

The principal hazard presented by exposure to radiation is cancer, caused by the induction of genetic mutations through the interaction of the radiation and cellular DNA. The potential health effects from ionizing radiation have been well studied

by various international and national organizations, notably UNSCEAR, ICRP, the (former) ACRP, the United Kingdom National Radiological Protection Board, the United States Environmental Protection Agency (U.S. EPA), and the U.S. NCRP, among others. As indicated previously, our knowledge of the health effects of ionizing radiation is largely based on epidemiological studies (e.g. see UNSCEAR, 2000).

People are exposed to radiation and radioactivity from sources external to the body and from radionuclides taken into the body through inhalation and ingestion; a more detailed discussion is provided in Section 5.0. For radiation protection purposes, a linear no-threshold dose-effect relationship is assumed, with coefficients derived from human epidemiology.

For **external gamma radiation**, a dose occurs when a person is exposed to a source of external gamma radiation. Thus, in concept, the product of the external gamma field and the time spent in that field determines the dose from external radiation.

On the other hand, for **internally deposited radionuclides**, the dose to body tissues can continue long after the intakes have ceased. This is very different from chemical exposure where such detailed models are not typically available. The ICRP has developed detailed models and methods for evaluating the time-integrated dose to body tissues[2] (e.g. ICRP, 1996).

To account for differences in the biological effect of equal absorbed doses arising from different types of radiation, the ICRP has developed weighting factors that convert the doses from different kinds of radiation to dose equivalents (i.e. doses with the same biological effects). In the absence of other modifying factors, this can be expressed as:

(2-1)

$$H = D \times Q$$

where:

H = dose equivalent
D = absorbed dose
Q = for alpha particles, Q = 20; for both beta particles and gamma radiation, Q = 1

The SI unit for absorbed dose and dose equivalent are Gray (Gy) and Sievert (Sv), respectively, and in environmental situations relevant to this document, doses would commonly be reported as mGy or mSv.

Finally, the ICRP has developed the concept of **effective dose** in which the doses to individual organs are multiplied by weighting factors (W_T) that reflect the relative radiosensitivity of the organ, and then the doses to all organs are added (Sv is the unit for effective dose). The concept is that exposure to 1 unit of effective dose (1 mSv, for example) causes the same risk as a uniform whole body irradiation from 1 mSv gamma radiation.

For exposure to the short-lived decay products of Rn-222, the traditional unit of working level (WL)[3] and working level month (WLM)[4] are used. Although the dose from short-lived radon decay products is almost entirely delivered to the lung (actually, the bronchial epithelial tissues), it is conventional to convert the dose from radon to a risk-equivalent effective dose using the conversion factor of 1 WLM = 4 mSv for members of the public and 1 WLM = 5 mSv for workers) (ICRP, 1977). Dose calculation procedures are discussed in more detail in Section 5.0.

Thus, for situations when people are exposed to different types of radiation, the overall effective dose can be estimated as:

(2-2)

$$\text{Effective dose} = ED_{external} + ED_{internal} + ED_{radon}$$

[2] For ionizing radiation, the risk (detriment in ICRP terms) for a member of the public is based on a 70-year integration of dose following the intake of a radionuclide. For example, if a person were to ingest "X" Bq radium-226 in a given year, a portion of the radium-226 would be eliminated from the body and a portion would be distributed among body tissues in accordance with well-established metabolic/biokinetic models; in addition, a portion would undergo radioactive decay. The portion of the radium-226 remaining in the body (and radioactive decay products) will continue to irradiate body tissues. The ICRP dosimetric models estimate the cumulative dose to body organs/tissues for 70 years following the year of intake. By convention, all of the dose estimated from the 70-year integration is assigned to the year of intake. This methodology is different from the use of the 75-year lifetime exposure used in chemical risk assessment; however, the use of the 70-year lifetime in radiological assessments is sufficiently similar for practical purposes.

[3] WL is defined as any combination of short-lived radon decay products that results in the ultimate emission of 1.3×10^5 meV of alpha energy.

[4] WLM is defined as exposure to 1 WL for 1 working month (170 h)

2.1.6 Dose to risk

The ICRP Publication 60 (ICRP, 1991) has established risk coefficients to convert dose to (lifetime) risk, these are:

- fatal cancer 5×10^{-5} per mSv
- non-fatal cancer 1×10^{-5} per mSv
- serious hereditary effects 1.3×10^{-5} per mSv
- total detriment 7.3×10^{-5} per mSv

Unless otherwise specified, all subsequent references to dose in this manual should be assumed to be the effective dose.

In Canada, it is not common practice in radiological risk assessments to calculate cancer risk and for licensed sites, annual doses predicted for members of the public are compared with a regulatory limit of 1 mSv. However, to allow for some harmonization between radiological and chemical risks under the FCSAP program, a dose-to-risk step has been included. This is discussed in Section 7.0.

2.2 Radiological Protection Concepts

In Canada, as elsewhere in the world, the approach to radiation protection follows that of the ICRP. In the ICRP Publication 2 (ICRP, 1959), the ICRP assumed that there was no threshold for genetic or carcinogenic effects of ionizing radiation, and hence that there should be no human-made exposure to ionizing radiation without some expectation of benefits. In the ICRP Publication 26 (ICRP, 1977), cancer was recognized as the main risk from exposure to ionizing radiation.[5] This publication recognized that different organs and tissues have different sensitivities to ionizing radiation; it presented the concept of effective dose (described earlier in this section), and recommended limits on maximum annual effective dose. The ICRP Publication 26 also recommended dose limits based on comparison of radiation risks with non-radiological risk that was considered widely acceptable by society.

The ICRP Publication 60 (ICRP, 1991) expanded on the framework for radiation protection and recommended the three basic principles, summarized as follows:

Justification – No practice involving exposures to radiation should be adopted unless it produces sufficient benefit to the exposed individuals or to society to offset the radiation detriment it causes.

Optimization – In relation to any particular source within a practice, the magnitude of individual doses, the number of people exposed, and the likelihood of incurring exposures should all be kept as low as reasonably achievable (ALARA principle), economic and social factors being taken into account.

Dose limitation – The exposure of individuals resulting from the combination of all the relevant practices should be subject to dose limits. The ICRP dose limits are set such that continued exposure at a dose just above the limit would be unacceptable on any reasonable basis.

The ICRP Publication 60 (ICRP, 1991) considered a number of factors including the concept of "acceptable" risk and variability in radiation dose from natural background in developing its recommended annual dose limit, which for a member of the public is 1 mSv. This publication also indicated that doses to members of the public should be kept well below the dose limit (ALARA). These principles have been adopted in this manual.

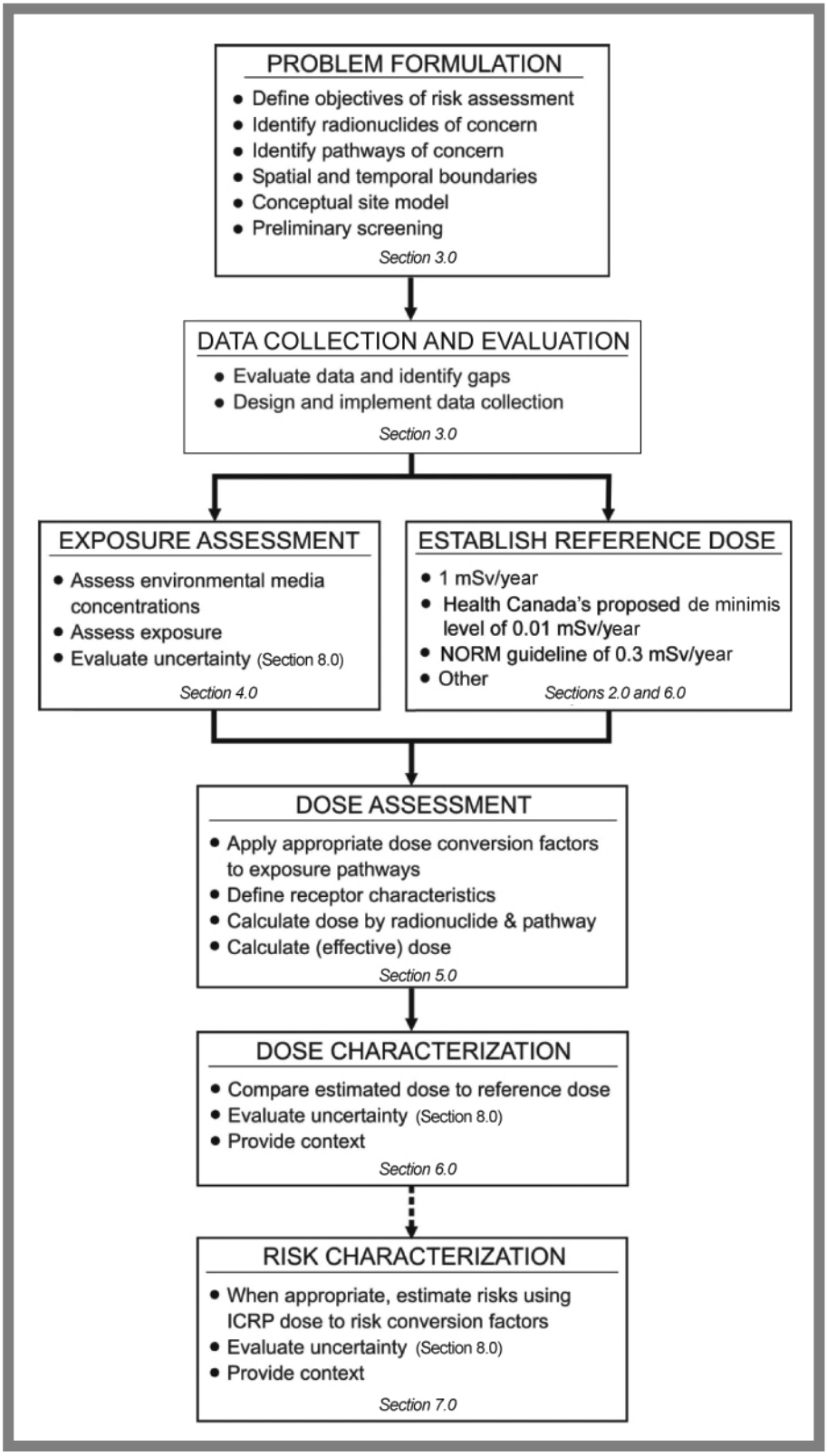
2.3 Framework

Although HHRA methods sometimes vary among different regulatory agencies, the frameworks for conducting the assessments are generally quite similar. Moreover, the frameworks for assessing radiological risks and chemical risks are quite similar. In Canada, the Health Canada guidance on HHRA (HC, 2004b; HC, 2010; HC, 2010b) is widely accepted and applied, and is similar to the U.S. EPA radiation risk assessment guidance (U.S. EPA, 1989, chapter 10)(http://www.epa.gov/oswer/riskassessment/ragsa/pdf/ch10.pdf). The framework for radiological risk assessment is outlined in Figure 2.5.

A project definition or problem formulation phase that clearly defines the objectives of the risk assessment should be the first step in the process. Subsequent steps in the risk assessment usually follow a step-by-step process similar to that shown in Figure 2.5. Each step is discussed in detail in the following sections.

[5] Radiation exposure has never been demonstrated to cause hereditary effects in a human population. Therefore, the most recent estimate of hereditary effects from exposure to ionizing radiation was based on spontaneous mutation rates of human genes and radiation-induced mutation of mouse genes (UNSCEAR 2001).

Figure 2.5 Steps in Human Health Risk Assessment for Radioactivity



2.4 Selection of Appropriate Reference Doses

In radiological risk assessments, calculated doses are compared with an appropriate reference dose. For licensed sites, the annual dose limit of 1 mSv was established by the CNSC. However, for the purpose of establishing dose objectives for use at federal sites contaminated with radionuclides, it is useful to look beyond the dose limits used by the CNSC. This section provides a brief overview of potentially applicable dose benchmarks. Some additional related discussion can be found in Section 6.0.

2.4.1 Canadian Nuclear Safety Commission dose limits

In Canada, the CNSC (AECB, 1987) specifies that at sites where annual doses **above background** are less than 0.05 mSv (50 μSv) the dose is acceptable, and it is not necessary to remediate the site further to reduce the dose. CNSC considers these sites as unlikely to cause significant adverse effects on human health.

The CNSC specifies that sites with doses less than 0.05 mSv/year above background do not need to undergo further remediation.

The CNSC has also established annual dose limits for members of the public at 1 mSv **above background** for sources they regulate, such as release (emission) limits for operating nuclear power stations and other CNSC-regulated sources.

The CNSC dose limit for members of the public at regulated sites is 1 mSv/year above background.

Many sources of radioactivity are not regulated by the CNSC – for example, federal contaminated sites, the radioactivity associated with phosphogypsum (a by-product of the manufacture of phosphate fertilizer), radioactivity from oil field wastes, rare-earths production, and other NORM activities.

2.4.2 Canadian guidelines for the management of naturally occurring radioactive materials

The development of regulations or guidelines for the management of NORM has gained increased attention in recent years. Because NORM radionuclides include those in the uranium and thorium decay series, the issues associated with the management of NORM wastes are therefore of potential relevance to contaminated sites.

NORM guidelines for unrestricted exposure are recommended as 0.3 mSv per year.

The *Canadian Guidelines for the Management of Naturally Occurring Radioactive Materials (NORM)* (HC, 2000) has recommended an annual dose limit of 0.3 mSv for unrestricted exposures. Essentially, the 0.3 mSv/year is a dose constraint to be applied to a single operation, to ensure that the combined doses from multiple sources do not exceed the public dose limit of 1 mSv/year. Similarly, 0.3 mSv/year is the value recommended by ICRP in such situations. In addition, the NORM guideline document also provides reference concentrations for unconditional release of diffuse NORM sources (large volume, low radioactivity): these reference concentrations provide a very efficient way to quickly evaluate a NORM contaminated site.

2.4.3 Decommissioning guidelines of the U.S. Nuclear Regulatory Commission

In July 1997, the U.S. Nuclear Regulatory Commission (U.S. NRC) issued its final decommissioning regulations governing the decommissioning of facilities subject to its licensing jurisdiction (U.S. NRC, 1997). Those regulations set clean-up criteria for sites contaminated with radioactive materials and outline a process for obtaining U.S. NRC approval of decommissioning plans.

The U.S. NRC dose criterion is 0.25 mSv per year excluding radon. The criterion is based on an assumed exposure to three or four sources.

The U.S. NRC rule uses a dose-based rather than a risk-based clean-up criterion. The annual dose criterion is 25 mrem (0.25 mSv) for unrestricted release, including the dose from groundwater sources of drinking water, but excluding the dose from radon and progeny. The limit is considered to apply to the average member of the critical group, where the critical group is consistent with ICRP terminology.

This criterion was developed in part from the generally accepted annual limit of 1 mSv for members of the public from all artificial sources, and the possibility of exposure to multiple sources. The U.S. NRC concluded that it was very unlikely that more than three or four sources would affect the critical group at any one time, and that a constraint of 25% or 30% of the 1 mSv/year limit appeared justified. It also concluded that presumed exposure to seven sources (as indicated by the U.S. EPA proposed annual clean-up limit of 0.15 mSv) was not supportable. The U.S. NRC considered that the 0.25 mSv/year criterion was reasonable from the standpoint of providing a sufficient and ample margin of safety for protection of public health and safety.

The annual dose from radon is **excluded** from the 0.25 mSv criterion. The U.S. NRC rationale for this exclusion is that the regulation should be based on criteria that are measurably distinguishable from

background radiation. Within buildings, wide variations in local concentrations of naturally occurring radon result in annual doses well in excess of 0.25 mSv. According to the U.S. NRC, it is very difficult to distinguish between naturally occurring radon and radon resulting from contaminated materials. It also notes that it is very difficult to predict prospective exposures to indoor radon owing to problems in predicting the design features of future building construction. For these reasons, the U.S. NRC contends "…. licensees will not be expected to demonstrate that radon from licensed activities is indistinguishable from background on a site-specific basis" (U.S. NRC, 1997, p.39083). Meeting the annual criterion of 0.25 mSv, **excluding the dose from radon**, for restricted release will be sufficient to comply with the regulation.

2.4.4 Radiation site clean-up guidelines of the U.S. Environmental Protection Agency

The U.S. EPA dose criterion is 0.15 mSv per year over natural background, excluding radon. This is based on an assumed exposure to seven sources.

On May 11, 1994, the U.S. EPA issued a working draft of radiation site clean-up regulations (U.S. EPA, 1994). The proposed regulations (which to date have not been finalized or promulgated) set standards for the remediation of soil, groundwater, surface water, and structures at federal facilities contaminated with radioactive material to allow these sites to be released for public use. The proposed regulations limit the annual doses received by members of the public to 0.15 mSv in excess of natural background levels for 1,000 years after completion of the clean-up. The annual 0.15 mSv limit **excludes** the dose from radon progeny. In addition, to be consistent with the maximum concentration limits for drinking water set under the U.S. EPA *Safe Drinking Water Act* (U.S. EPA, 1996a) (http://water.epa.gov/lawsregs/rulesregs/sdwa/), the regulations require that annual exposures from groundwater be no greater than 0.04 mSv.

According to the U.S. EPA, the annual dose limit of 0.15 mSv corresponds to a lifetime excess cancer risk of less than 3×10^{-4} over a 30-year exposure. In other rulemakings, for example related to the use of phosphogypsum containing NORM, this risk value has been considered to be "presumptively safe" by the U.S. EPA (1992). With this proposal, the U.S. EPA was intending to protect the reasonably maximally exposed (RME) individual in the population located on or near a previously contaminated site that has been released for public use after undergoing remediation. The RME is defined by the U.S. EPA as the individual receiving the radiation exposure experienced by the 95th percentile and above of the population at a released site. Most of the exposed individuals will receive much less exposure than that experienced by the RME.

Doses from radon are **excluded** from the clean-up criteria because all existing and future buildings on the remediated sites must meet the guidelines of the U.S. EPA radon program (i.e. 4 pCi/L or 148 Bq/m^3); that is, the remedial actions should be designed to limit radon progeny concentrations to less than 0.02 WL, or approximately 148 Bq/m^3 of radon. Using the U.S. EPA average indoor occupancy factor of 0.6 (U.S. EPA, 1994) and the ICRP dose conversion factor (DCF) of 4 mSv/WLM for members of the public (ICRP, 1993), the 0.02 WL limit converts to a dose of approximately 2.5 mSv/year.

2.4.5 Health Canada proposed "essentially negligible" dose for contaminated sites

Health Canada proposes an "essentially negligible" dose of 0.01 mSv/year above background and exclusive of radon for federal contaminated sites. This level is consistent with the ICRP (2004) recommendation of an annual dose rate of 0.01 mSv as the "minimum constraint" that should be considered for application in any situation. Additionally, this is consistent with the incremental lifetime cancer risks that are considered to be negligible for genotoxic carcinogens. It is recognized that the determination of 0.01 mSv above background levels may not be possible to achieve or even to quantify at many sites, and at certain sites, a level up to the NORM guideline of 0.3 mSv may be required. Health Canada advises that the ALARA principle be used at sites where the dose is above 0.01 mSv.

2.5 Risks from Chemicals Versus Ionizing Radiation

The dose assessment portion of radiological risk assessment has been developed within a somewhat different framework to that for chemicals. Chemical risk assessment involving carcinogens is generally based on laboratory experiments with animals exposed to relatively high doses or concentrations, whereas the risks for radiation and radioactivity are based primarily on human epidemiology.

In radiological risk assessments, radiation external to the body (primarily gamma radiation) can be an important contributor to dose and risk. There is no analogue for external radiation in chemical risk assessment. In assessing doses and risks from ionizing radiation, it is necessary to add the doses from external and internal sources of radiation and convert them to a common metric **effective dose** (measured in Sv), where the total risk of all the combined exposures is proportional to the effective dose.

Effective dose considers the different radionuclides and/or radiation types, the total individual dose to each organ (based on metabolic models), and the relative susceptibility (i.e. cancer risk) of each organ to that dose. The organ doses include those doses received during exposure, as well as those committed[6] because of the ongoing irradiation caused by internally deposited radionuclides. Then, as described earlier in this section, the organ doses are multiplied by weighting factors that reflect the relative radiosensitivity of the various organs, and summed to give the total effective dose.

Within the regulatory framework for risk assessment, there are several conflicting regulatory policies for limiting routine exposures of the public to radionuclides and other chemical carcinogens (e.g. Overy and Richardson, 1995). In particular, there are regulatory inconsistencies in the level of accepted health risks associated with radionuclides (e.g. Kocher and Hoffman, 1991). A perception also exists that generally, in North America, radiation exposure is not regulated as stringently as chemical exposure (Travis et al., 1989). However, a more recent analysis of risk management practices for radiation and chemicals has determined that risk management strategies for both ionizing radiation and carcinogenic chemicals are well developed and similar in principle, and that risk management strategies for both provide a high degree of health and environmental protection (HC/AECB, 1998).

Because radionuclides and chemical carcinogens often coexist, it is important to consider differences in risk assessment and risk management, and to establish a practical approach to defining the level of exposure that is applied for regulation and/or risk management. One problem is whether to and how to combine risks from a given exposure to a variety of chemical and radiological hazards into a single number to support a decision regarding clean-up. Risk factors for chemicals are uncertain and are assumed to be conservative (biased high). If used as they are presented for a single contaminant, there can be quite a high degree of conservatism. When several contaminants are concurrently evaluated, the conservatism has potential to become compounded (assuming independence among the risk factors), leading to highly skewed results that are inappropriate for cost-benefit analysis (Pollock et al., 1995). Notwithstanding this observation, evidence from Canadian federal contaminated sites (Franz Environmental Inc., 2005) shows that the probability of co-occurrence of multiple carcinogenic chemicals at any one site is low. So this point, namely simultaneous exposure to multiple chemical carcinogens, is effectively moot with respect to federal sites. However, it is almost always the case that sites contaminated with radionuclides will also contain one or more chemical carcinogens.

A 1998 report by Health Canada and the AECB (HC/AECB, 1998) provides a comprehensive overview of the approaches to risk assessment and risk management of cancer risks from chemicals and ionizing radiation. This joint report attempted to provide an overview of the methods and approaches for assessing and managing exposure to ionizing radiation and carcinogenic chemicals, and prepared a summary table (HC/AECB, 1998, table 5) showing what they considered to be important similarities and differences. It is recognized that this report is somewhat dated, but the comparisons provided in the table are still valid to date; the report table has been revised and updated to reflect the current situation (Table 2.2).

[6] For ionizing radiation, the risk (detriment in ICRP terms) for a member of the public is based on a 70-year integration of dose following the intake of a radionuclide. For example, if a person were to ingest “X” Bq Ra-226 in a given year, a portion of the Ra-226 would be eliminated from the body and a portion would be distributed among body tissues in accordance with well-established metabolic/biokinetic models; in addition, a portion would undergo radioactive decay. The portion of the Ra-226 remaining in the body (and radioactive decay products) will continue to irradiate body tissues. The ICRP dosimetric models estimate the cumulative dose to body organs/tissues for 70 years following the year of intake. By convention, all of the dose estimated from the 70-year integration is assigned to the year of intake. This methodology is different from the use of the 75-year lifetime exposure used in chemical risk assessments; however, the use of the 70-year lifetime in radiological assessments is similar for practical purposes.

Table 2.2 Comparison of Risk Assessment and Management Aspects for Ionizing Radiation and Carcinogenic Chemicals

Risk Assessment	Ionizing Radiation	Carcinogenic Chemicals
Sources of contaminants	Natural and artificial	Natural and artificial
Number of potential carcinogens	Relatively stable and known	Number continues to increase with expanding toxicological database
Types of effects at environmental exposure levels	Long-term carcinogenic effects	Long-term carcinogenic effects
	Hereditary effects (evidence equivocal, but potential risks calculated)	Other non-cancer effects, such as immunological hypersensitivity
Sources of risk data	Primarily epidemiological studies on humans	Both toxicological studies on animals and epidemiological data
Risk assessment approach	Total risk assessed using a single unifying approach	Separate risk derived for different agents
Risk extrapolation	Linear no-threshold extrapolation from high-dose data Some evidence for practical threshold effects for specific radionuclides	Linear no-threshold extrapolation from high-dose data for genotoxic carcinogens Evidence of threshold effects for non-genotoxic carcinogens and possibly also for some genotoxic carcinogens
Risk estimates	Includes risk of fatal cancer, plus an allowance for non-fatal cancers weighted for severity of type and ease of curing, length of life lost or impaired, risk of serious hereditary disorders	Generally focus on cancer incidence (fatal and non-fatal) Different types of cancer are generally treated equally, without weighting. Different chemicals have different potencies.
Uncertainties in risk estimates	Generally less uncertainty due to reliance on human data	Less uncertainty where more data exist Generally greater uncertainty due to reliance on animal data for many chemicals Lack of human data for many chemicals
Goal	To minimize risk, recognizing economic and social factors	
Sources of recommended exposure limits	Internationally harmonized recommended system of radiation protection Limits generally set as low as reasonably achievable, social and economic factors being considered	De minimis risk variably defined between 1 in 10,000 (10^{-4}) and 1 in 1,000,000 (10^{-6}) International recommendations not generally harmonized; often different recommendations from national and international organizations Limits for carcinogens in drinking water may vary 10,000-fold in theoretical cancer risk.
Principles for controlling exposure	Limit based on acceptable risk, and variations in unavoidable natural background radiation Limit based on human-health considerations	Limits for individual carcinogens aim for a lifetime risk of 10^{-4} to 10^{-6} with the limit dependent on best-available technology economically achievable, background levels, etc. Individual limits are not compared against the total background of natural carcinogens. Although human health is generally the critical factor, limits are sometimes based on ecological considerations.

Risk Assessment	Ionizing Radiation	Carcinogenic Chemicals
	Actual exposures to be maintained as low as reasonably achievable, with economic and social factors taken into consideration Limit covers all exposures from all regulated practices Total risk of all health effects readily calculated on the basis of international recommendations	Limits apply to individual chemicals, often via only one route of exposure. No attempt to calculate total risk associated with all individual limits
Manner of implementation	Dose limitation: public dose limits lower than occupational limits Optimization of risk-benefit Control at source for regulated practices and at point-of-use	
Public dose limits for industry	Operational limits for nuclear generating facilities based on achievable levels, significantly lower than the legal dose limit	Operational limits for industry based on achievable levels, and similar to actual exposure levels
Risk-benefit approach	Although applied inconsistently, given significant consideration in the optimization of health protection A monetary limit on cost of preventing a premature death by reduction in radiation exposures from industrial sources has been recommended by the AECB.	Traditionally given lesser consideration, although is being used under the *Canadian Environmental Protection Act*, and in context of best-available technology economically feasible No set limit on cost of preventing a premature death by reduction in exposure to chemical carcinogens

Source: HC/AECB, 1998.

The 1998 report by Health Canada and the AECB included, among others, the following conclusions:

- "Risk management strategies for both ionizing radiation and genotoxic chemicals are also well-developed and are similar in that they both set legal limits to exposures, endorse the ALARA principle, and employ approaches such as source controls, point-of-use controls, and education" (p. 41).

- "There is a lack of consensus regarding levels of risk acceptability for ionizing radiation or genotoxic chemical hazards. Rather, the *acceptable* levels of risk associated with established guidelines vary up to a million-fold" (p. 41).

- "The Joint Working Group finds that the risk management strategies for regulated practices for both ionizing radiation and genotoxic chemicals provide a high degree of health protection. It is not possible to determine whether environmental exposures to ionizing radiation or genotoxic chemicals pose the greater risk of cancer at this time" (p. 42).

2.6 Moving Forward

It should be noted that many aspects of components of radiological risk assessment will be adopted within the Health Canada chemical risk assessment guidance and vice versa to effect revisions to existing standards of practice in that discipline. Standards of practice for radiological risk assessment will not be the only procedures to be revised.

These steps, although minor, will significantly increase the harmonization between radiological and chemical risk assessment in Canada. These steps will also significantly improve the ability of risk assessors to communicate the results of radiological risk assessments to affected communities; such communities often do not comprehend why current radiological risk assessment appears to provide less health protection than chemical risk assessment.

3.0 PROBLEM FORMULATION

The problem formulation step is perhaps the most crucial step in the risk assessment process. This step is where the risk assessor makes decisions as to what will be considered in the risk assessment. This step is also crucial in terms of helping to identify and evaluate potential remedial actions that will be considered. Thus, the goals of the risk assessment are important. Also at this step, the determination is made if the assessment will be a screening type, deterministic assessment, or a more detailed probabilistic assessment.

3.1 Define the Objective of the Assessment

Problem Formulation

- Define objectives for the assessment.
- Preliminary evaluation of available data and identify data gaps.
- Develop conceptual site and exposure pathways model.

The first step is to have a preliminary review of all available data for the site and to develop objectives that will guide the development of the (preliminary) conceptual site model that:

- identifies radionuclides of concern
- identifies and characterizes pathways of concern
- establishes reference doses or alternative guidelines
- generally serves as the basis for a preliminary screening

Often at the problem formulation stage, data may be scarce. A preliminary screening risk assessment that intentionally uses assumptions likely to provide (reasonably) conservative estimates of dose can be used to eliminate radionuclides and pathways that are trivial; this can help in focusing on further effects of data collection and analysis.

A clear definition of the objectives of the assessment is crucial before proceeding. In particular, experience has shown that it is essential to clearly define the land use(s) relevant to the assessment. This may be the existing land use, the use intended upon redevelopment/remediation, or both.

Risk assessments that will be used for the development of remedial action (target) levels may need to consider a wide range of potential exposure conditions. Risk assessments that are being conducted to demonstrate regulatory compliance may have very specific requirements regarding the exposure and dose assessment.

3.2 Selection of Deterministic or Probabilistic Assessment

A deterministic risk assessment is the first step in the radiological assessment process.

The deterministic approach to an assessment involves the use of a single value for each variable in the exposure equation. In general, these values are set to be cautious (conservative) and are believed to provide an estimate of exposure and effect that does not underestimate actual conditions. As indicated previously, in many instances, the initial screening assessment should be carried out in a conservative manner in which the model parameters selected and the assumptions made ensure that the estimated dose/risk is near conceivable (reasonable) worst case. In many instances, it is useful to perform a sensitivity analysis to identify the parameters that have the greatest influence on the predicted doses and risks. Where it is desirable to more fully characterize and quantify the effects of uncertainty, or where the frequency of different dose or risk levels across a population are desired, a probabilistic analysis should be considered.

The risk assessor should consider the sources and likely effects of uncertainty on the deterministic assessment. In general terms, uncertainty arises from lack of knowledge (e.g. limited soil characterization data) and from chemical and biological variability that arises from the heterogeneity, such as that arising from sampling point to sampling point variation in concentration or soil characteristics, and person-to-person variation in the amount of air they breathe or the amount of fish they consume. As a minimum, the risk assessment should provide a qualitative discussion of the sources of uncertainty concerning data, assumptions, and parameters that might affect the outcome of the risk assessment, such as the factors shown in Table 3.1.

In carrying out a radiological risk assessment at a federal contaminated site, it is recommended that the risk assessor carry out a deterministic assessment first. This will assist the assessor in determining whether an additional study is necessary and, if so, to help focus such a study – likely a probabilistic assessment. In the event the need for a probabilistic assessment is indicated, the assessor must identify if sufficient data are available to support such an assessment. It should also be noted that risk assessments are carried out in an iterative fashion, starting with a conservative screening level assessment and, as needed, evolving toward increasingly realistic assessments (see Section 3.4).

Table 3.1 Examples of Factors to Consider in Examining Sources of Uncertainty in Human Health Risk Assessment

Receptors	• age group (infant, child, teen, adult, whole life) • activity group (subsistence hunter/gatherer, fishing camp operation, residential versus industrial land use, etc.) • average versus reasonable maximum exposure
Potential exposure pathway	• inhalation • ingestion • dermal (chemical HHRA only, except tritium) • external gamma
Temporal scale	• past, present, future

Dose/risk assessment endpoints	• effective dose
	• risk of fatal cancer
	• dose/risk statistics
	  ○ mean, confidence interval of mean
	  ○ percentile of distribution (e.g. 90%, 95%)
	  ○ reasonable worst case

Data quality	• adequacy of site data • modelling uncertainties • lack of representativeness (e.g. If a hunter/gatherer does not consume water from the site, then the amount of water consumed is not relevant.)

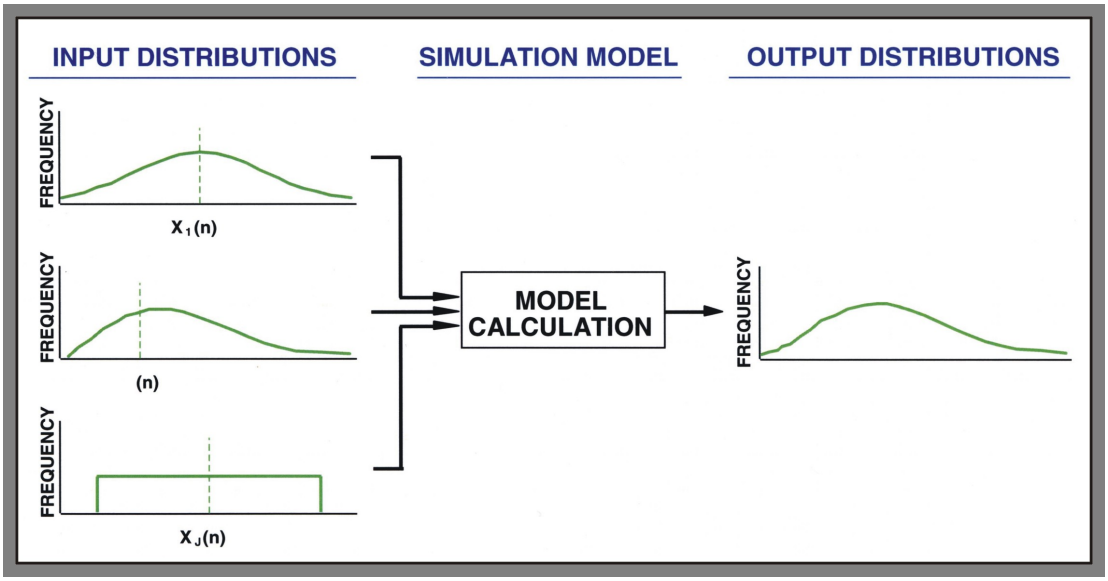
Note: Many FCSAP sites are at remote northern locations, and a residential scenario is highly unlikely. In such a situation, the need to consider a residential scenario would be screened from further consideration.

A probabilistic risk assessment is used only at a contaminated site if the conservative deterministic assessment indicates a potential problem.

When initial screening calculations indicate a potential problem, then a probabilistic assessment may be appropriate. A probabilistic assessment can also be used to develop best estimates and statements of confidence in the results, as well as frequency distributions of dose/risk across a population. In a probabilistic risk assessment, the uncertainty of key input parameters is explicitly and quantitatively included by defining a probability distribution function (PDF) for each parameter. These PDFs represent the range and frequency (likelihood) of values for a parameter, either across a population, within a set of measured data, or reflect the limited knowledge concerning a particular variable. A computer simulation is performed through repetitive random selection of input parameters from the respective PDFs using Monte Carlo statistical sampling techniques and executing the model algorithms. Figure 3.1 shows conceptually how multiple and different probability distributions can be used to derive an output, which is then represented by a unique output distribution.

Even when full probabilistic analysis is performed, deterministic estimates of exposures, doses, and risks should be reported to facilitate comparisons with the probabilistic analysis and to assist in risk communication.

Figure 3.1 Schematic of Probability Distribution Functions



Health Canada has provided specific guidance on conducting Monte Carlo assessments in Appendix B of of Health Canada's *Federal Contaminated Site Risk Assessment in Canada, Part V: Guidance on Human Health Detailed Quantitative Risk Assessment for Chemicals ($DQRA_{Chem}$)* (HC, 2010b); this document should be consulted in the event that a probabilistic assessment is necessary. Additional information is available in literature, for example U.S. EPA (1997).

Models to complete a probabilistic assessment are available. Software, originally designed to analyze uncertainty in business applications, can be useful to complete a probabilistic assessment of environmental risk. Two widely used programs that use models of a spreadsheet form are Crystal Ball® (www.decisioneering.com) and @Risk (www.palisade.com). Some fate, transport, and dose models, e.g. RESRAD (U.S. DOE and U.S. NRC, 2004) and UTAP (SENES, 1987), include probabilistic modelling capabilities not only for transport but exposure. Probabilistic models specifically for deriving estimated radiation doses have also been formulated. However, the risk assessor must be familiar with these models before using them in a probabilistic assessment.

3.3 Conceptual Site Model

Conceptual models are used in risk assessment to delineate the focus of the assessment. The conceptual model illustrates all environmental elements and processes that are of relevance to the exposure of the defined group of individuals that could potentially be exposed at the site, and is crucial to the risk assessment process. The conceptual model identifies all relevant emissions sources, environmental fate and transport processes, and exposure pathways to the defined group of individuals (receptors). Human exposure pathways frequently include environmental exposures through direct environmental exposures to media, such as soil and water. Possible indirect exposure includes ingestion of wildlife, garden vegetables, and fish, depending on the specifics related to the site. An example of a conceptual site model is presented in Table 3.2. The following sections provide a discussion on some of the major inputs to the conceptual site model that the risk assessor should consider.

The conceptual site model is required at the outset of the risk assessment.

Table 3.2 Example of a Conceptual Site Model

Hypothetical mine site

Are there any human health issues/concerns related to past activities at the site based on preliminary site investigations?

Conceptual model

Preliminary quantitative risk assessment using maximum concentrations and conservative assumptions

Data availability

Preliminary data available for U-238, Th-230 and Th-228, Ra-226, and Po-210 in surface water, soils, sediments, and external gamma radiation at site

Receptors that may be considered

- camper on site who would obtain drinking water, plants, fish, small and large game from site and who may be exposed to impacted soils directly or via wind-blown dust
- site inspector who would be present on site for 2 days per year; only exposed to external gamma radiation, soils, and tailings

Pathways of exposure

CAMPER

- inhalation
- external gamma
- incidental ingestion of soil
- ingestion of water
- ingestion of plants
- ingestion of fish
- ingestion of small game
- ingestion of large game

SITE INSPECTOR

- incidental ingestion of soil
- external gamma ray exposure
- incidental inhalation of dust

What other data are needed to conduct the assessment?

- air concentrations
- edible plant concentrations
- fish concentrations
- small game concentrations
- large game concentrations
- groundwater concentrations to model impacts to fish-bearing waters if contaminant plume has not yet reached the surface water

How to infill the data?

Simple fate and transfer spreadsheet models using generic transfer factors (for example):

- accumulation in plants from soil
- accumulation in fish from water and/or sediments' accumulation in game from soil/plants/ water/air
- levels in air (Rn-222 and resuspended soil particulate)

3.3.1 Source characterization

A review of the historical uses of the site, as well as available monitoring data should provide the basis for the selection of radionuclides to be included in the assessment. At the simplest level, the concentrations found in environmental media can be compared with dose or risk-based guidelines, such as those developed by the *Health Canada Guidelines for Canadian Drinking Water Quality* or the U.S. NCRP surface soil screening limits (U.S. NCRP, 1999), to determine which radionuclides are above screening levels and should be assessed further.

Available information should be compiled for concentrations of radionuclides in various media (e.g. soil, groundwater, surface water, biota). Possible sources of information, additional to any data from site investigations that may have been carried out, include, among others, open literature information from various federal agencies (e.g. Geological Survey of Canada); research papers and studies carried out by government agencies, universities, and private companies; and scientific articles in published journals.

In some instances, environmental concentrations may vary considerably from season to season or even day to day (e.g. ambient radon). Other media environmental concentrations are expected to be relatively constant (e.g. concentrations of long-lived radionuclides in soil).

In addition to radionuclide concentrations in environmental media, some data on the (radiation) exposure rate will often be available from site measurements. Gamma surveys provide important information with respect to external radiation exposure. Exposure rates at radioactive contaminated sites have typically been reported in units of microroentgen per hour (µR/h), which in SI units is 0.01 µGy/h. These measurements should also be considered in the assessment. Seasonal differences in gamma exposure rates are expected to be small, except in locations where a thick snow cover might provide shielding, with the amount of shielding depending on snow pack density. Gamma radiation levels can vary spatially and, unless there was a reason to assign a preference to use a particular area, it would be reasonable to use a mean gamma exposure rate for the purpose of dose assessment.

The risk assessor should give preference to measured data where valid measurement data are available; however, there are instances where measured data are not available. In such cases, environmental fate and transfer models can be used to develop the appropriate media concentrations or to supplement site measurement. In some instances, empirical models derived from measured data or from combinations of measured and literature data can be useful to infill missing or inadequate data. The risk assessor needs to determine if the level of assessment requires a site-sampling program to be developed for augmenting limited available data, or if the use of environmental fate and transfer models and their inherent uncertainties are suitable for the purpose of the assessment. In the case of screening-level assessments, the use of environmental fate and transfer models is often appropriate and routine.

As with all human health assessments (both radiological and chemical), it is important for the risk assessor to delineate the extent of contamination. This includes consideration of the spatial distribution, both surface area and depth. In some circumstances, there may be potential for radionuclides to migrate away from a source; examples include radioactivity in surface streams or Rn-222 released to the atmosphere. In such cases, it may be necessary to consider, via measurement data, or models, or a combination of data and models, the correlation of radionuclides downstream in water, or sediment for a release to a stream or river, or dispersion in the atmosphere in the case of radon.

3.3.2 Environmental pathways

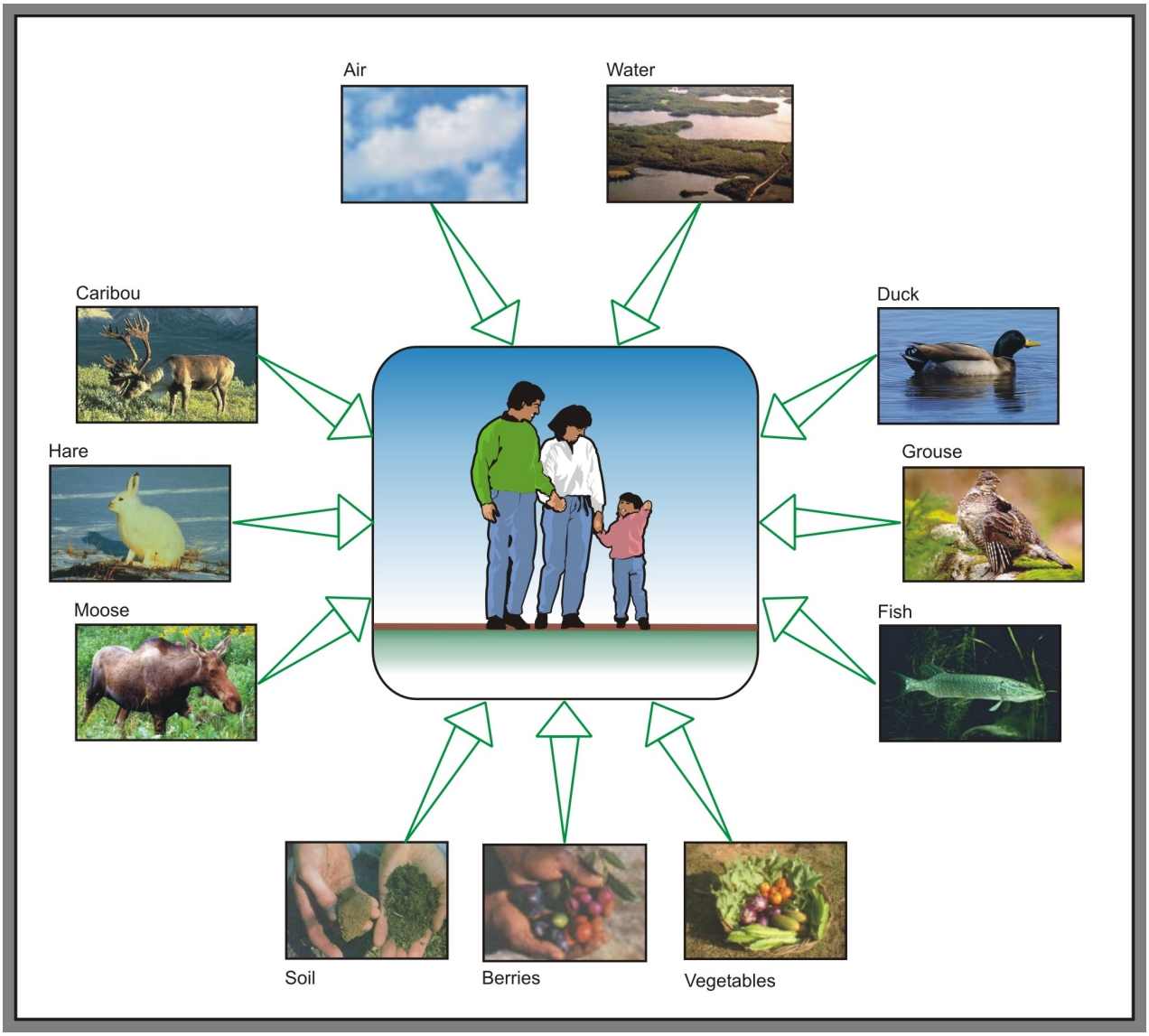
The specific pathways of exposure to humans that should be considered for inclusion in an assessment include:

- consumption of drinking water
- consumption of fish
- consumption of terrestrial vegetation (e.g. above-ground and below-ground plants, berries)
- consumption of wild game (e.g. caribou, rabbit, water fowl)
- inhalation of radionuclides, particularly radon
- inadvertent ingestion of soil
- external gamma radiation

In developing the conceptual model, the risk assessor needs to determine which of these pathways is operable at the site. An example of a pathways diagram is included in Figure 3.2. The specific pathways that are appropriate and that need to be included will be specific for each site.

Some radionuclides require specific considerations. For example for radon, the health effects arise from exposure to radon decay products and the primary exposure pathway is the migration of radon into a building and subsequent inhalation of radon decay products.

Figure 3.2 Exposure Pathways to Be Considered in the Radiological Risk Assessment



3.3.3 Exposure scenarios

Risk assessments are conducted for a defined group of individuals, referred to as "receptors." Different groups of receptors may be present on a site, and the relevant exposure pathways, receptor characteristics, and receptor behaviours must be defined for each group of receptors. The risk assessor needs to select the appropriate receptor. Because values reflecting the exact characteristics and behaviours are never available for each individual in a receptor group, generalizations and assumptions are made using professional judgment to select appropriate values for each group. Recommended default receptor characteristics for radiological risk assessment are presented in Appendix B. Frequently in contaminated site radiological risk assessment under the FCSAP program, receptor groups may include intruders to the site, workers present in an on-site building, residents living off-site but nearby, and farmers, fishers, and hunters pursuing activities near or on the site. In addition, First Nations people and Inuit who use the site for traditional activities should also be considered as receptors, particularly for northern and/or remote sites. In the case of uranium mining sites, the CNSC will establish a prohibition against living on the site, but it is important to consider exposure arising from casual access such as by a hunter or trespasser accessing the site. The risk assessor needs to select the appropriate receptors from the range of possibilities.

Each group may have different but overlapping exposure pathways. Residents living nearby may be exposed via inhalation pathways and via local produce from their backyard gardens, but will not be exposed via drinking water if their water supply is provided by a municipal drinking water system. Farmers may obtain drinking water from a well drawing from impacted groundwater and also be exposed via inhalation and via ingestion of produce that they harvest. Different receptor characteristics can be used to represent physiological features of each receptor group. For example, the risk assessor may wish to choose a higher inhalation rate for the working farmer than for the resident living nearby. Furthermore, each receptor group may demonstrate different behaviours that will affect exposure, such as time spent outdoors, and frequency and duration of exposure at the site. For example, workers may be present on site 5 days a week and 48 weeks of the year, whereas hunters and fishers may be present on or near the site only for a limited time. The risk assessor must define all relevant features of each receptor group in order to define all exposure scenarios.

Issues concerning the pathways analysis of non-volatile particulate radionuclide releases into the environment are generally similar to corresponding releases of non-radioactive or chemical species. The risk assessor may need to consider the use of an appropriate model for estimating concentrations in different media if direct measurements are not available (see Section 4.0 for a discussion of models).

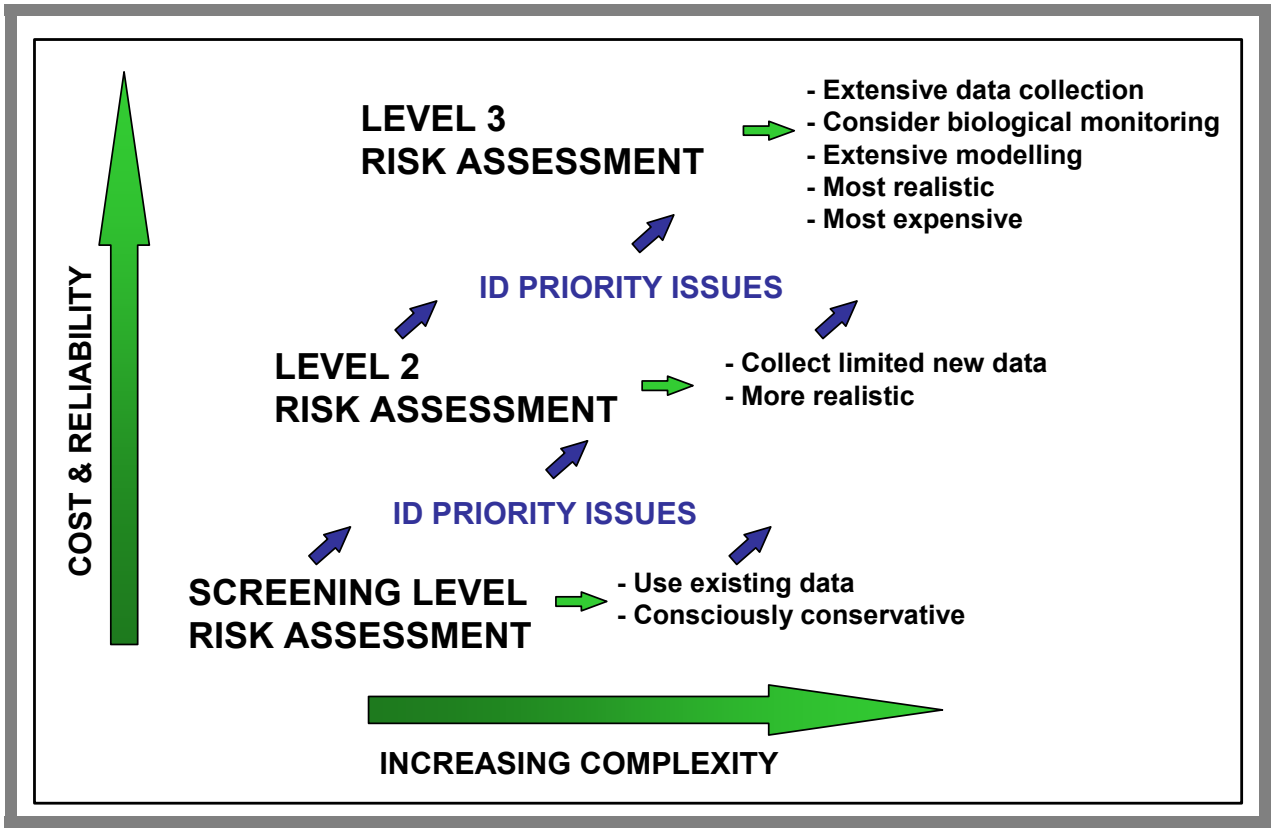
3.3.4 Receptor characterization

Receptor characterization typically includes the following features of each receptor group: body weight, inhalation rate, soil ingestion rate, drinking water intake rate, and food ingestion rates. Skin surface area, for estimation of dermal exposures, is not required for radiological risk assessment, unless tritium is measured at the site. Furthermore, receptor characterization includes how much time each visit or activity lasts, unless looking at tritium, on the contaminated site (or in critical areas of the site) in order to quantify exposure frequency and duration for each exposure scenario. Appendix B provides recommended single-point and probabilistic functions related to receptor characteristics that the risk assessor should use in the radiological risk assessment, Section 5.0 provides more details on receptor characterization.

3.4 Iterative Nature of Dose and Risk Assessment

Dose and risk assessment can be carried out at different levels of detail and complexity. It is normal to start with an intentionally conservative screening-level assessment that integrates available site data infilled (augmented) as necessary with generic assumptions and/or with the application of models to predict unmeasured parameters. Such analyses are designed to ensure, to the extent that knowledge and data permit, that actual doses and risks are more likely than not to be overestimated. Hence, if the doses to receptors estimated through such a conservative screen are sufficiently below the reference dose/risk criteria, then it may not be necessary to collect more data and perform more detailed assessment. On the other hand, if the doses (risks) from a conservative screening exceed the predetermined level of dose (risk), then a further data collection phase and more detailed risk assessment may be warranted. The risk assessment is refined through an iterative process, such as that illustrated in Figure 3.3, until the uncertainties in the dose and risk assessment are judged adequately within available resources.

Figure 3.3 Staged Approach to Risk Assessment



Note: ID = identify

4.0 FATE AND TRANSPORT MODELLING

An overview of available fate and transport, and exposure models that the risk assessor can use in radiological risk assessments at federal sites is provided in Section 4.1. Environmental transport pathways potentially relevant to radionuclide emissions to air from a contaminated site are described in Section 4.2, and potentially relevant pathways for environmental transport of radionuclide emissions to surface water and to groundwater from a contaminated site are described in sections 4.3 and 4.4, respectively. Potentially relevant pathways for environmental transfer of radionuclides through soil, plants, and animals are described in sections 4.5, 4.6, and 4.7, respectively. Finally, potential exposure pathways are described in Section 4.8. In Section 4.9, the special cases of exposures to gamma radiation and radon decay progeny in a home located on a contaminated site are discussed, and other special radionuclides (tritium and C-14) are discussed in Section 4.10.

4.1 Overview of Available Fate and Transport Models

For some applications, it may be important for the risk assessor to consider exposure through pathways for which site-specific information is very limited or non-existent. It may also be desirable for the risk assessor to predict levels in the environment at some distance from the site or at some time in the future (if no remedial action is taken, for example), and/or after some type of change (remediation), such as in the source concentration. In these circumstances, fate and transport modelling may be the only means of estimating exposures. This type of modelling allows an estimation of the level of radioactivity in various environmental compartments based on the mobility of specific radionuclides. Figure 4.1 provides an illustration of fate and transport processes that should be considered in the risk assessment.

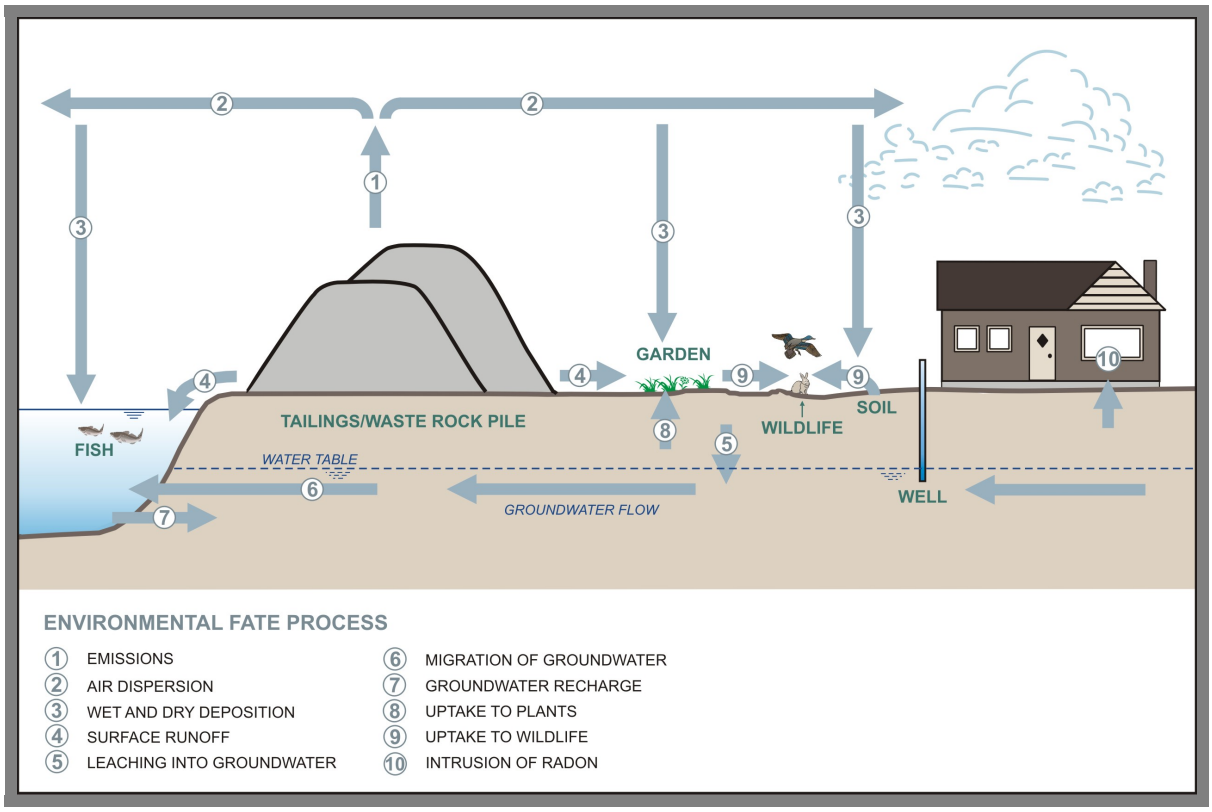
Fate and transport modelling can be a valuable tool for:

- predicting the importance of specific pathways
- predicting overall exposures from all applicable multi-media exposure pathways in the absence of actual data
- predicting the impact of mitigation measures or the impact of no mitigation
- prioritizing additional data requirements

Models range from simple to complex, and the data requirements are contingent on the model selected.

Many of the radionuclides, uranium and thorium for example, behave in the environment like trace metals. The environmental behaviour of radon, tritium, and C-14 differs from other radionuclides, and therefore their environmental transfer and exposure pathways are generally modelled in a different manner.

Figure 4.1 Schematic of Fate and Transport Processes Considered in a Radiological Risk Assessment



It may be important for the risk assessor to consider radioactive decay and ingrowth, especially if the assessment is looking at some future time. For some radionuclides, such as natural uranium and thorium, it is not necessary to include ingrowth and decay because the effects on the activity inventories are likely to be inconsequential when one considers the long half-lives of these radionuclides. However, radioactive ingrowth and decay are potentially very important for other radionuclides, such as Ra-226, Pb-210, and Po-210. In some instances, the concept of radioactive decay and ingrowth can be used to assist the assessor in making reasonable assumptions about levels of radionuclides that might be present in various media but for which there are no data. For example, if the assessor has information on the concentrations of Pb-210 in soil or sediment, it might be reasonable to assume that Po-210, the decay product of Pb-210, is present at the same concentration as the result of ingrowth.

A large number of fate and transport models have been developed around the world and are available for risk assessors to consider; however as discussed previously, many of these models are quite complex with considerable input data requirements. Very often, data, at least at the screening stage, are limited, and simple conservative models are often more appropriate to use. A summary of some of the more common models that have been used in radiological risk assessments is provided in Table 4.1.

Table 4.1 Examples of Selected Fate and Transport Models

	CANADIAN MODELS
CSA N288.1	Canadian Standards Association model for calculating release limits for radioactive material in airborne and liquid effluents from normal operation of nuclear facilities (CSA, 1987)
CSA N288.2	Canadian Standards Association model for calculating radiation doses to the public from a release of airborne radioactive material under hypothetical accident conditions in nuclear reactors (CSA, 1991)
UTAP	The Uranium Tailings Assessment Program for calculating source terms, multimedia exposure pathways, and radiation doses to the public from uranium mining, milling operations, and related tailing piles (SENES, 1987)
	U.S. AND OTHER MODELS
Reg Guide 1.109	Regulatory Guide for calculating annual doses to members of the public from routine releases of reactor effluents for the purpose of evaluating compliance to regulations (U.S. NRC, 1977)
Reg Guide 3.51	Regulatory Guide for calculating annual doses to members of the public from airborne radioactive materials resulting from uranium milling operations (U.S. NRC, 1982)
RESRAD	Residual Radioactivity manual for evaluating radioactively contaminated sites (Yu et al., 2001) http://web.ead.anl.gov/resrad/home2/
PRESTO	Prediction of Radiological Effects due to Shallow Trench Operations guide for low-level radioactive waste sites to evaluate radiation exposure from contaminated soil layers (U.S. EPA, 2000) http://www.epa.gov/radiation/assessment/presto.html
MEPAS	Multimedia Environmental Pollutant Assessment System includes pathways and scenarios for radioactive and chemical hazardous materials (Buck et al., 1995) http://mepas.pnl.gov/earth/mepasmain.html
GENII	Generation II Model for Environmental Dose Calculations for estimating potential radiation doses from routine and accident releases of radionuclides to air or water and residual contamination from spills or decontamination operations (Leigh et al., 1993) www.rsicc.ornl.gov/rsiccnew/cfdocs/qryPackageDownload.cfm
NUREG/ CR-5512	U.S. Nuclear Regulatory Commission model for analyzing exposure pathways for critical groups at decommissioned facilities (often referred to as the D and D Code) (U.S. NRC, 1992) http://www.nrc.gov/

A detailed comparison of such models is beyond the scope of this manual, but others have made such evaluations (e.g. U.S. EPA, 1996a; U.S. DOE, 1994). In addition, several international studies have provided an opportunity for large-scale testing of assessment models against field data, as well as for inter-comparison of results and modelling approaches among investigators from many countries. The IAEA has sponsored the ongoing Validation of Environmental Model Predictions. Some international programs include:

- BIOMOVS (sponsored by the Swedish Radiation Protection Agency from 1986–1990)
- BIOMOVS II (sponsored by organizations in Sweden, Canada, and Spain from 1990–1996)
- BIOMASS (organized by the IAEA in 1966)

4.1.1 Selection of an appropriate model

The selection of the appropriate model will be specific for each risk assessment application. The objectives of the study, the level of analysis and complexity required, the amount of available information, and how confident the assessor (and other stakeholders) want to be are all important factors in the choice of a model. For simple screening-level risk assessments and for simple radiological assessments at contaminated sites, the risk assessor should use simple models for performing calculations before proceeding to more detailed models. The complexity of the model should be consistent with the complexity of the overall risk assessment in which the model is being used; therefore, screening-level assessments should employ simple models.

The risk assessor is encouraged to start with a simple model, using assumptions likely to produce reasonable upper-bound estimates of risk before progressing to more detailed data-intense models.

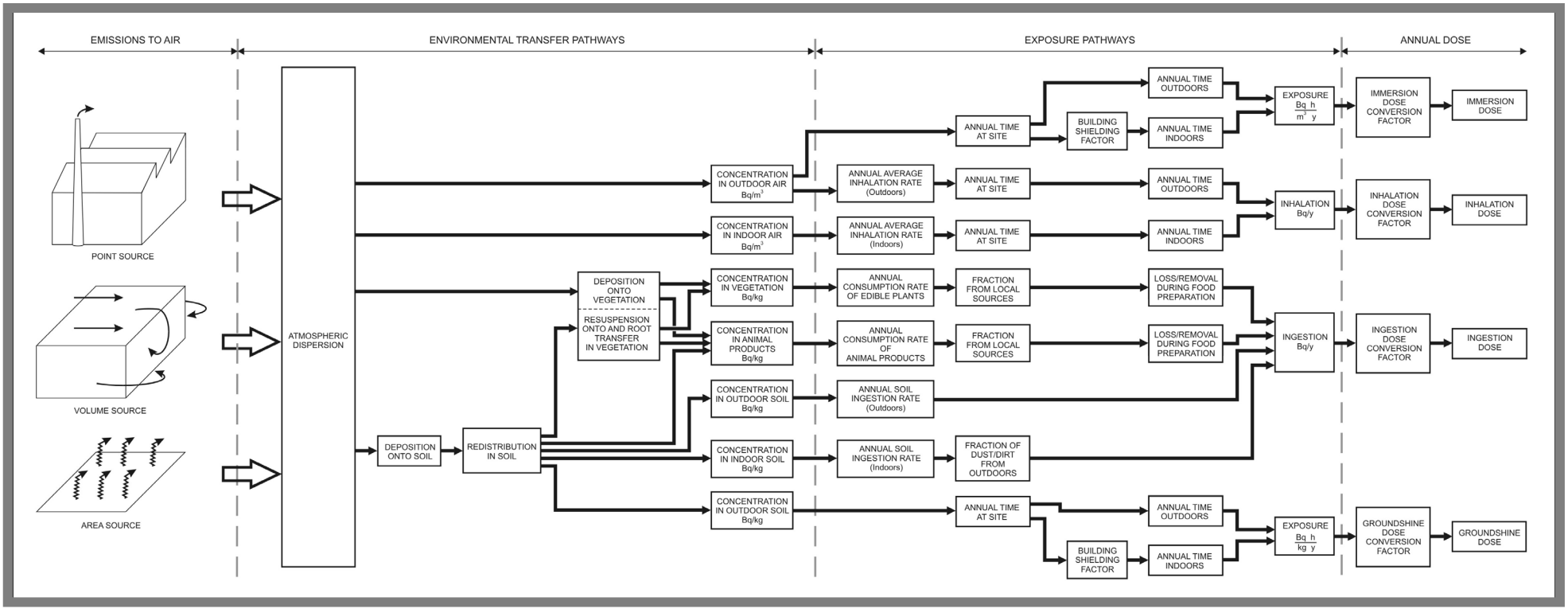
As noted earlier and discussed in Section 8.0, probabilistic models are now more routinely used to assess the effect of uncertainty in the predicted environmental concentration, doses, and risk. However, it is suggested that the starting point of assessment be a conservative screening-level assessment with progression to more detailed and complex assessment (including probabilistic analysis) in a staged iterative approach, as required. The need for progression to greater complexity and detail will be determined as the results from each stage of assessment are evaluated in consideration of the objectives set out in the problem formulation stage (see Section 3.0).

The following section provides an overview of some of the models commonly used for radiological risk assessment. The presentation is not intended to be comprehensive, and considerable reference is made to open literature publication. The users of the manual are cautioned to ensure that the model(s) they adopt for a particular site and scenario are both appropriate and reasonable for the intended application and available data.

4.2 Environmental Fate and Transport – Releases to Air

Radiation exposures from radioactivity releases to the atmosphere may be attributed to environmental transfer and exposure pathways represented in Figure 4.2. The risk assessor can use the mathematical models described in this section to predict radioactivity release rates into the air from selected sources (see 4.2.1) and atmospheric dispersion of radioactivity to receptor locations.

Figure 4.2 Schematic Representation of Environmental Transfer and Exposure Pathways for Emissions to Air



Note: y = year

4.2.1 Source terms

The amount of dust generated from a contaminated site and the rate at which radon is released to the air from a site are very dependent on a number of factors, including the extent of contamination, whether contamination is near surface or at depth, the extent of surface cover (landscaping/vegetation, asphalt, concrete, building[s], etc.), vehicular traffic or other dust-generating activities (agriculture, earth moving, etc.), and local meteorological conditions (especially precipitation that inhibits resuspension of soil particles) – all of which should be considered in the conceptual site model.

In the absence of site-specific measurements of suspended respirable dust levels, appropriate factors for wind erosion from the U.S. EPA publication AP-42 (U.S. EPA, 1995b) are readily applicable for use to estimate dust emission source terms that would then be multiplied by the radioactivity content to arrive at estimates of airborne radioactive dust emissions. For screening-level risk assessments, a simple assumed aerially suspended respirable (≤10 µm) dust level of 0.76 µg/m^3 may be assumed if there are no vehicles travelling on unpaved roads or other construction activity (HC, 2004b). Levelton Consultants Ltd. (2005) has developed a screening model for evaluation of fugitive dust generation at simple sites.

UNSCEAR (2000) provides a succinct discussion concerning sources of radon and the processes that affect the release of radon from soils. One of the key parameters that controls radon transport in soils is the radon diffusion coefficient. Because diffusion occurs through the unsaturated pore space of the soil, the diffusion of radon in soil, where the soil is compacted or the pore space is filled with water (saturated), will be very much slower than in uncompacted or unsaturated soils.

A number of authors report models for estimating the radon flux from the surface of porous media (e.g. soil or waste) including, among others, UNSCEAR (2000), U.S. NRC (1980), and U.S. EPA (1983).

The radon flux for any planar surface can be estimated by scaling (approximately) on the basis of radium content and exposed surface area and assuming a radon flux of 1 Bq (Rn-222) m^{-2} s^{-1} per Bq (Ra-226) g^{-1}.

For dry soils, using the methods and values reported by the U.S. NRC, a unit area radon flux of about 1 Bq m^{-2} s^{-1} per Bq g^{-1} is estimated. In its analysis of uranium mill tailings, the U.S. NRC adapted a nominal radon flux of 1 Bq Rn-222 m^{-2} s^{-1} per Bq Ra-226 g^{-1} (U.S. NRC, 1980), which is consistent with the above value. Thus for screening level assessments, a nominal radon flux of 1 Bq (Rn-222) m^{-2} s^{-1} per Bq (Ra-226) g^{-1} can be assumed for exposed surfaces of contaminated soils or wastes. Radon transfer from soil into buildings is described in Section 4.9.2.

4.2.2 Atmospheric dispersion

The objectives of atmospheric dispersion calculations are to estimate radionuclide concentrations in air, notionally at breathing height above the ground surface for the following conditions:

- annual average concentrations in outdoor air at varying distances from continuous long-term releases at a constant rate (such as from sites left unremediated)
- short-term concentrations in outdoor air from short-term releases (such as during temporally finite remedial activities)
- releases from point, area, or volume sources

4.2.2.1 Continuous long-term release at a constant rate

Radioactivity releases to the atmosphere are subject to dilution in air and are dispersed by the prevailing winds. The following discussion applies to releases that occur at a constant rate and are continuous over long periods of time (i.e. weeks to months).

The annual average radionuclide concentration in air at a receptor location in the range 0.1 km to 20 km from a point source at a height above the ground can be conservatively estimated using long-term average dilution factors for typical Canadian weather and uniform wind rose shown in Figure 4.3. The annual average concentration of radioactivity in air at *x* metres from the release point can be calculated using the equation:

(4.1)

$$C_x = K_h \quad R \qquad \text{Bq/m}^3$$

where:

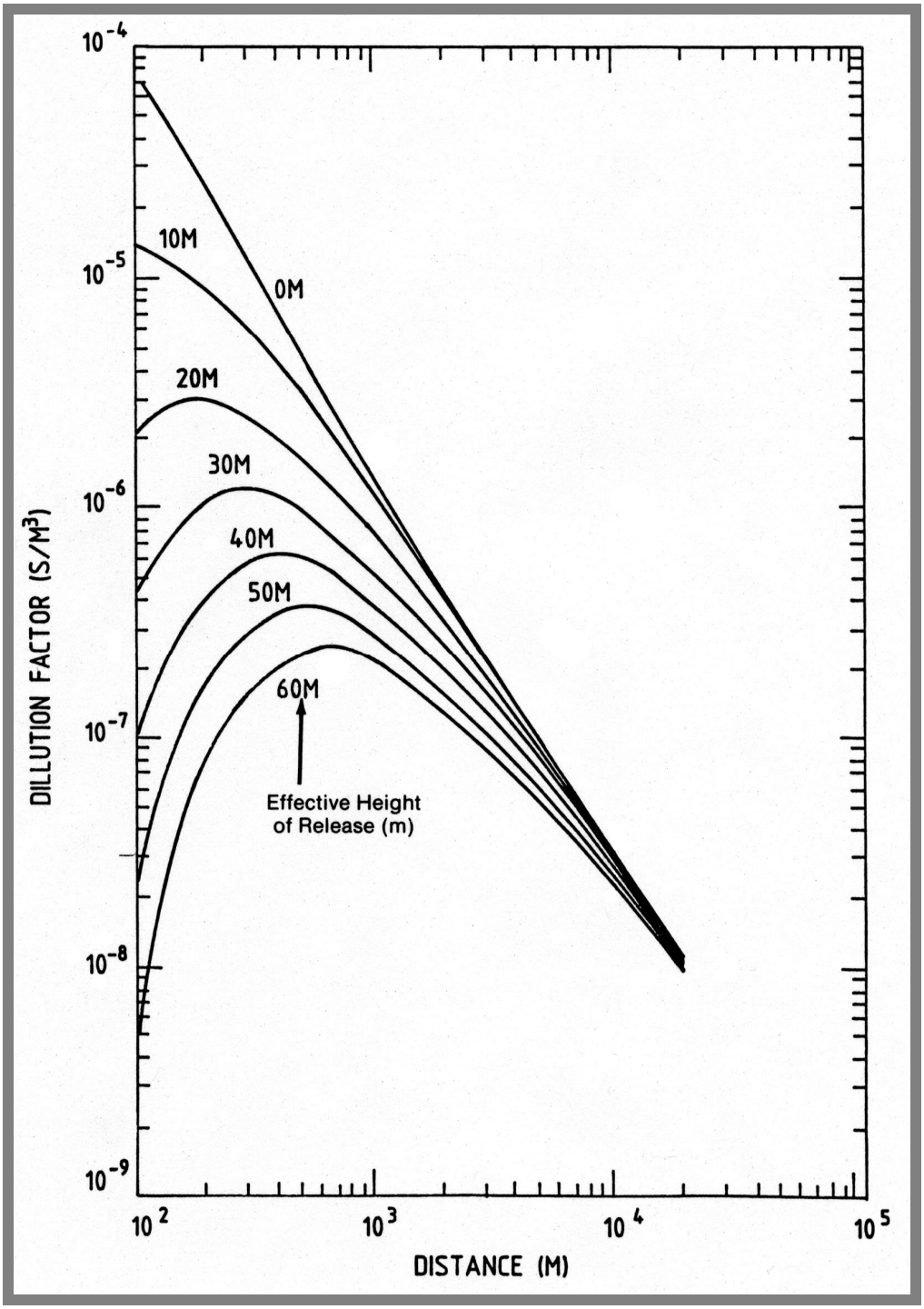
C_x = annual average radionuclide concentration at distance *x* metres from release point
K_h = dilution factor for a release point *h* metres above the ground and at a distance *x* metres from the receptor (Figure 4.3), s/m^3
R = annual average release rate from the source, Bq/s

It should be noted that the location of the highest annual average concentration in air to which receptors at ground level are exposed is a function of the release height. Receptors exposed to ground-level releases experience the highest concentrations at the closest accessible locations to the release point. However, receptors exposed to elevated releases experience the highest concentrations at separations

For screening purposes, the curve shown in Figure 4.3 is recommended for dilution factors.

from the release point that increase with increasing release height. For example, at a release height of 20 m, the highest ground-level concentration is at a source/receptor separation of approximately 200 m, whereas at a release height of 0 m, the highest ground-level concentration is at the minimum source/receptor separation of approximately 100 m.

Figure 4.3 Long-Term Average Dilution Factors for Typical Canadian Weather and Uniform Wind Rose



Source: Modified from CSA, 1987, fig. A1.

A useful model that has been used by many assessors to estimate radon concentrations in air near area sources is the Schiager Box Model (Schiager, 1974), described in Appendix A (Section A1.0).

More complex models

Where possible, the risk assessor is encouraged to use actual measurement data to assess radiological exposure. Where models are necessary, their complexity should be consistent with the scope and complexity of the risk assessment; simple models should be used for screening-level risk assessments. However, real-world scenarios may be more complex for one or more of the following reasons:

- The radionuclide release rate varies over time (diurnal or seasonal variations).
- The plume deposition is expected to significantly affect the concentration.
- The receptor separation is greater than 20 km from the source.
- The wind rose, atmospheric stability category, and wind speed are not uniform among all compass directions.
- The source configuration is complex.

For dispersion under more complex scenarios, the risk assessor can use models that require more input data (e.g. wind speed, direction and stability class, ground roughness) and more computing power. The U.S. EPA regulatory model ISC3 (U.S. EPA, 1995a) has been validated in many applications and widely used for these purposes, and is convenient to use for this type of application. Levelton Consultants Ltd. (2005) has designed a model to estimate fugitive dust emissions from contaminated sites.

4.2.2.2 Short-term and prolonged-term releases

It is not expected that the risk assessor will be required to evaluate exposures for acute (short-term) releases of radioactivity from a federal site contaminated with radioactive substances. However, should the need arise, the risk assessor is directed to the CSA (1991) for guidance to calculate the average radionuclide concentration in air, for the short-term duration of interest, at a range 0.1 km to 60 km from a point source. The standard describes dispersion models applicable to short-term and prolonged-term releases of radioactivity (i.e. durations of 3 minutes up to 1 hour and of 1 hour up to 1 day, respectively). The radionuclide concentrations in air from more complex source geometries, release scenarios, and ground contours can be calculated using computer models such as the U.S. EPA (1995b).

(4.3)

$$f_e = 1.0 - 0.0479\ e^{-u/(4.39\,x)} - 2.1963\ e^{-u/(38.6\,x)} + 1.2442\ e^{-u/(28.4\,x)}$$

where:

f_e	=	radon progeny equilibrium ratio, no units
u	=	average wind speed in the area, m/s
4.39, 38.6, 28.4	=	constants, 1/s
x	=	downwind distance from source to receptor, m

4.2.2.3 Significant radioactive decay or ingrowth during dispersion

Radioactive decay during dispersion by the wind

During atmospheric dispersion of short-lived radioisotopes (relative to the travel time from source to receptor), there can be significant decrease in the radionuclide concentrations in air from radioactive decay in addition to the decrease in concentration from dispersion. The decrease in concentration can be divided into two multiplicative factors – one for dispersion and one for radioactive decay:

(4.2)

$$C_x = K_h \quad R \quad e^{-\lambda t} \qquad \text{Bq/m}^3$$

where:

C_x	=	radionuclide concentration at distance x metres from the source, Bq/m^3
K_h	=	dilution factor for a release point h metres above the ground and at a distance x metres from the receptor (Figure 4.3), s/m^3
R	=	annual average release rate from the source, Bq/s
λ	=	radionuclide decay constant, 1/s
t	=	travel time from source to receptor (separation x, metres, divided by average wind speed in m/s), s

The dilution factor (K_h) can be extracted directly from the graph in Figure 4.3, whereas the default values for average wind speed in Canadian cities can be obtained from the Canadian Climate Normals Data (www.canadainfolink.ca/climate.htm). The radionuclide decay constants for the uranium series radionuclides are provided in Appendix A (Section A2.0).

Ingrowth of radon decay progeny

The ingrowth of radon decay progeny during the dispersion of radon gas from a release point to a receptor can be calculated using the model recommended by the U.S. EPA (1986). The radon progeny equilibrium ratio (ranging from 0 to 1) can be calculated as a function of distance from the source using the equation 4.3 below.

Site-specific wind speed and direction data should be used where available. However, use of data from the nearest weather station may be acceptable, particularly for screening-level risk assessments.

Ingrowth of thoron decay progeny

The ingrowth of thoron decay progeny during the dispersion of thoron gas from a release point to a receptor can be calculated using the Bateman equations (Evans, 1955); average wind speed and source-receptor separation using the equation 4.4 below.

The dilution factor (K_h) can be extracted directly from the graph in Figure 4.3, based on distance to the receptor. Site-specific wind speed and direction data, if available, are preferable to any assumptions. However, use of data from the nearest weather station may be acceptable, particularly for screening-level risk assessments.

4.3 Environmental Fate and Transport – Releases to Surface Water

Radiation exposures from radioactivity releases to surface water may be attributed to environmental transfer and exposure pathways indicated in Figure 4.4. Mathematical models described in this section can be used to calculate annual average radionuclide concentrations in impacted water (e.g. contaminated soil).

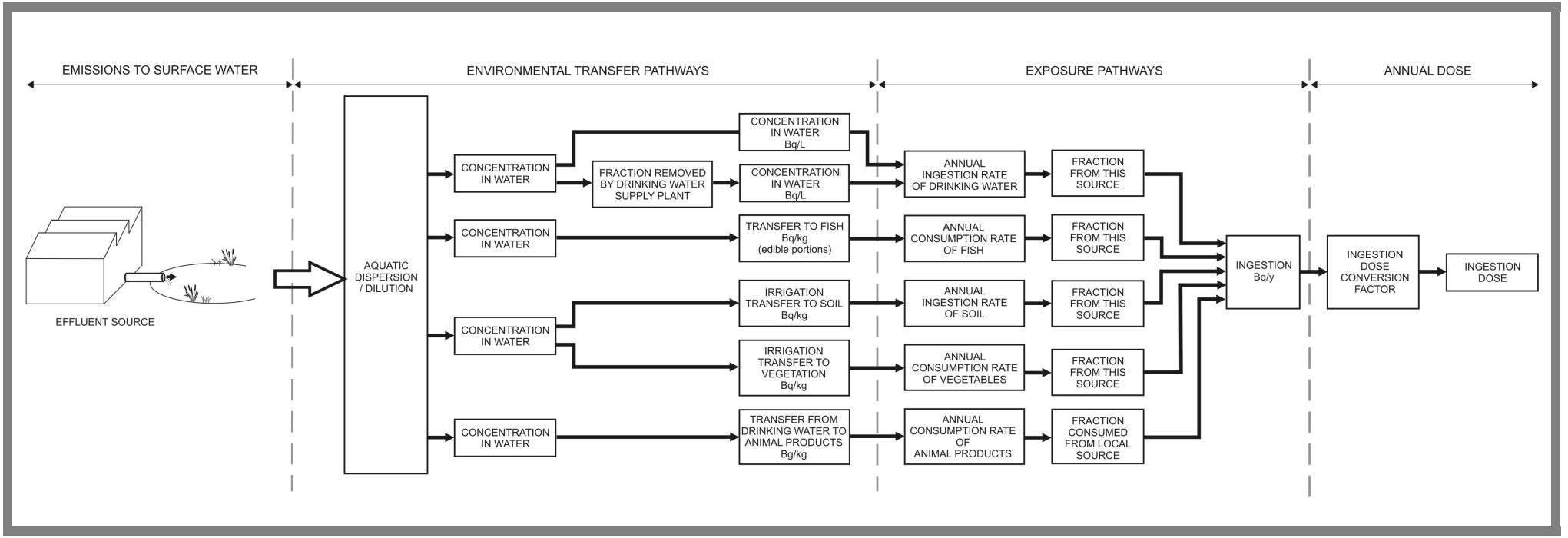
(4.4)

$$C_{212} = K_h \; R_{220} \; \lambda_{212}(1 - e^{-\lambda 220\, t}) / \lambda_{220} \qquad \text{Bq/m}^3$$

where:

C_{212} = Pb-212 concentration at a distance x metres from the thoron source, Bq/m^3
K_h = dilution factor for a release point h metres above the ground and at a distance x metres from the receptor (Figure 4.3), s/m^3
R_{220} = annual average release rate of Rn-220 from the source, Bq/s
λ_{212} = Pb-212 decay constant, 1/s
λ_{220} = Rn-220 decay constant, 1/s
t = travel time from source to receptor (separation x metres, divided by average wind speed in m/s), s

Figure 4.4 Schematic Representation of Environmnetal Transfer and Exposure Pathways for Emissions to Surface Water



Note: y = year

4.3.1 Dispersion in surface water

Radioactivity released to surface waters of rivers or lakes may be diluted by local currents and removed by sedimentation. In the absence of reliable data for removal by sedimentation, the CSA (1987) recommends the following dispersion model:

(4.5)

$$C_w = R_r \ \beta \ e^{-\lambda T} / (D_f \ Q_v) \qquad \text{Bq/L}$$

where:

C_w = annual average radionuclide concentration in water at the point of use, Bq/L
R_r = annual average release rate, Bq/s
β = effluent recirculation factor, no units
λ = radioactive decay constant, 1/s
T = transport time from point of discharge to point of use, s
D_f = dilution ratio, the quotient of the annual average effluent concentration at the point of discharge to the annual average concentration at the point of use, neglecting radioactive decay, no units
Q_v = annual average seepage discharge rate, L/s

Liquid dispersion in large bodies of water is highly complex and dependent on site-specific characteristics such as bottom topography, wind-induced currents, seasonal variation of heating or cooling of lake waters, and mass exchange between near-shore and off-shore waters.

CCME (2005) recommends a default value for the dilution ratio (D_f) of 10 for discharges of groundwater into a lake or river (surface water). Alternate values (higher or lower) may be appropriate on a site-specific basis, and may be determined by modelling, site-specific studies, or both.

4.3.2 Deposition/build-up in sediment and on shoreline

4.3.2.1 Deposition in sediment

Radioactivity in surface water will adsorb onto particulate matter and accumulate in sediments over time. The radioactivity concentration in sediments under equilibrium conditions with constant concentrations in water can be estimated using the following equation (CSA, 1987, section 6.7):

(4.6)

$$C_{sed} = C_{sw} \ k_d \qquad \text{Bq/kg}$$

where:

C_{sed} = resulting radioactivity concentration in sediment, Bq/kg (dry sediment)
C_{sw} = radioactivity concentration in surface water, Bq/L
k_d = water-to-sediment distribution factors, L/kg (dry sediment)

Recommended values for water-to-sediment distribution factors are provided in Appendix A (Section A3.0). It should be noted that the distribution factors provided in Section A3.0 must be converted before using equation (4.6).

4.3.2.2 Deposition on shoreline

The radionuclide concentration in shoreline sediments can be estimated using the following expression:

(4.7)

$$C_{is} = k_d \ C_{sw} \ [1\text{-}e^{(-\lambda i \ tb)}] \ \text{Bq/kg}$$

where:

C_{is} = concentration of radionuclide *i* in sediment, Bq/kg
k_d = water-to-sediment distribution factors, L/kg (dry sediment)
C_{sw} = concentration of a radionuclide in water adjacent to the sediment, Bq/L
λ_i = decay constant of radionuclide *i*, 1/h
t_b = length of time the sediment is exposed to the contaminated water, h

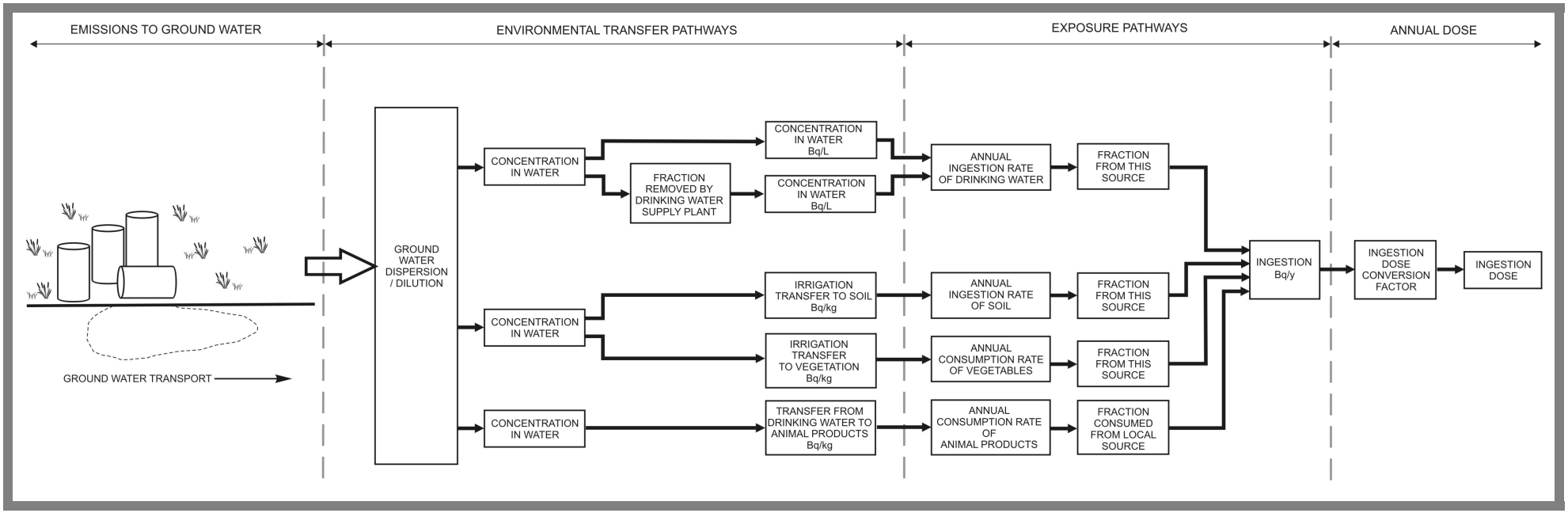
Recommended values for water-to-sediment distribution factors (Section A3.0) and radionuclide decay constants (Section A2.0) are provided in Appendix A. The length of time the sediment is exposed to the contaminated water (corresponding to the duration of the release) is a site-specific value that must be assigned by the risk assessor. In the absence of site-specific measurements, the value in the square brackets of equation (4.7) should be conservatively taken as unity (CSA, 1987).

4.4 Environmental Fate and Transport – Releases to Groundwater

Radiation exposures from radioactivity releases to groundwater may be attributed to environmental transfer and exposure pathways indicated in Figure 4.5. Mathematical models described in this section can be used to calculate annual radionuclide intake rates via ingestion, dermal absorption (for tritium), and annual exposure rates from groundshine (contaminated soil).

It is strongly recommended that radionuclide concentrations in groundwater be estimated by modelling as a last resort only. All reasonable efforts should be made to collect and analyze groundwater samples from the site for the purpose of estimating exposures. Should modelling be attempted, only models suited and calibrated to the site should be considered. It is beyond the scope of this manual to recommend specific models.

Figure 4.5 Schematic Representation of Environmental Transfer and Exposure Pathways for Emissions to Groundwater



Note: y = year

4.5 Fate and Transport in Soil

Environmental site assessments of federal contaminated sites will always include sampling of soil for contamination. Therefore, the need to predict soil concentrations will never be required. Health Canada will not entertain or review a risk assessment for a federal contaminated site where contaminants in the soil (and often also in the groundwater) have not been directly measured.

The material that follows is included for completeness only, and for those very rare occasions when it might be necessary to predict radioactivity levels in soil at some off-site location, following transport of radiologically contaminated soil particulate matter by wind or precipitation runoff.

4.5.1 Undisturbed soil

The risk assessor should use the following equation for computing radionuclide concentrations on the ground from the calculated airborne particulate concentrations arising directly from on-site sources (not including air concentrations resulting from resuspension) (U.S. NRC, 1977, p. 8). First, the deposition rate of radionuclide *i* is calculated, using the following relationship:

(4.8)

$$D_{di} = \sum_{p} C_{adip} V_p \qquad \text{Bq/(m}^2\text{ s)}$$

where:

D_{di} = resulting direct deposition rate of radionuclide *i*, Bq/(m^2 s)
C_{adip} = calculated direct air concentration of radionuclide *i* in particle size *p*, Bq/m^3
V_p = deposition velocity of particle size *p*, m/s

The default value for the deposition velocities is provided in Appendix A (Section A2.0).

The concentration of radionuclide *i* on the ground surface due to constant deposition at the rate D_{di} over time interval *t* is obtained from:

(4.9)

$$C_{gi}(t) = D_{di}\left[\frac{1 - e^{[-(\lambda_e)t]}}{\lambda_e}\right] \qquad \text{Bq/m}^2$$

where:

$C_{gi}(t)$ = calculated ground surface concentration of radionuclide *i* at time *t*, Bq/m^2
t = time interval over which deposition has occurred, s
D_{di} = resulting direct deposition rate of radionuclide *i*, Bq/(m^{2}s)
λ_e = effective removal rate constant, 1/s

The effective removal constant from the soil λ_e, is governed by two processes: (1) radioactive decay, characterized by the decay constant, λ_r, and (2) environmental loss, characterized by a physical removal constant, λ_s, where $\lambda_e = \lambda_r + \lambda_s$. The constant λ_s is taken to be 2.20×10^{-10} s^{-1} for all radionuclides. This corresponds to an upper limit of the physical half-life for removal from the soil of about 100 years (CSA, 1987, section 5.3.2). These relationships are assumed to apply to all radionuclides deposited on the ground. As mentioned previously, the radionuclide decay constants are provided in Appendix A (Section A2.0).

4.5.2 Tilled soil

Routine tilling of soil mixes the deposited radioactivity to the tillage depth. The radioactivity concentration is assumed to be uniform throughout the tillage depth, and the concentration at time *t* can be calculated using the following equation (CSA, 1987, section 5.3.1):

(4.10)

$$C_{vs} = C_{gi}(t) / (d\,\rho) \qquad \text{Bq/kg}$$

where:

C_{vs} = radionuclide concentration in vegetated soil to the tillage depth at time *t*, Bq/kg
$C_{gi}(t)$ = calculated ground surface concentration of radionuclide *i* at time t, Bq/m^2
d = tillage depth, m
ρ = dry soil density, kg/m^3

Recommended values for tillage depth and dry soil bulk density are 15 cm and 1,600 kg/m^3, respectively (CSA, 1987). CCME (2000) recommends default dry bulk densities of 1.4 g/cm^3 for fine-grained soil, and 1.7 g/cm^3 for coarse-grained soil (based on soil particle density of 2.65 g/cm^3, and porosities of 0.47 and 0.36 for fine- and coarse-grained soils, respectively.

Under some scenarios, the radionuclide behaviour in soil may be more complex than characterized by the simple models described above. The risk assessor is encouraged to ensure that the complexity of the models used is consistent with the scope and complexity of the overall risk assessment approach. Should it be appropriate to apply more complex models to predict deposition and build-up in soil, the risk assessor is directed to the D and D Code (U.S. NRC, 1992), ISC3 (U.S. EPA, 1995a), PRESTO (U.S. EPA, 2000), and RESRAD (Yu et al., 2001) models.

4.5.3 Deposition/build-up in soil (irrigation)

Radioactivity in irrigation water applied to vegetable gardens and lawns will accumulate in the soil over time. The concentration of a radionuclide in soil from irrigation under steady-state conditions can be estimated using the following relationship (CSA, 1987, section 6.3):

(4.11)

$$C_{s,irr} = C_{sw} \times \frac{L'}{\lambda_g} \times \frac{1}{d \times \rho} \quad \text{Bq/kg}$$

where:

$C_{s,irr}$ = the resulting concentration of a radionuclide in soil through irrigation with contaminated water, Bq/kg
C_{sw} = radionuclide concentration in irrigation water, Bq/m^3
L' = annual average irrigation rate (default value of 2.3 × 10^{-8} recommended by CSA, 1987), m^3/(s m^2)
λ_g = effective removal constant from the soil, 1/s
d = tillage depth, m
ρ = dry soil density, kg/m^3

Note that $\lambda_g = \lambda_r + \lambda_s$ where λ_r = radioactive decay constant and $\lambda_s = 2.20 \times 10^{10}$ s^{-1}, corresponding to a removal half-life from the soil of 100 years.

4.6 FATE AND TRANSPORT TO VEGETATION

4.6.1 Deposition to vegetation

One of the mechanisms of radioactivity transfer to vegetation is by deposition of radionuclides in the air. The vegetation concentration through deposition of non-volatile radionuclides from the plume can be calculated using the following equation (U.S. NRC, 1982, section 1.3) using the equation 4.12 below.

The effective removal constant from forage and crops, (λ_e), is determined by radioactive decay (λ_r) and physical removal processes such as wind, rain, and plant growth (λ_p), and is given by $\lambda_e = \lambda_p + \lambda_r$. The physical removal constant is taken to be 5.73 $\times 10^{-7}$ 1/s, which corresponds to a half-life for removal of 14 days. The deposition time(t_e) is taken as 30 days for pasture grass and 50 days for all other crops. The default values for the deposition velocity and yield densities for different types of vegetation are provided in Appendix A (Section A2.0).

4.6.2 Root transfer from soil to vegetation

Radioactivity in root-zone soil transfers into vegetative and reproductive plant tissues throughout the growing season. Root uptake of radionuclides incorporated into surface horizons of soil can be parameterized by the transfer factors B_v and B_r, representing the ratio of the elemental concentrations in plant and soil at harvest. The parameters B_v and B_r (Baes et al., 1984) are given by:

(4.12)

$$C_{vi} = C_{air\ total} \times V_p \times F_r \times E_v \times \frac{\left[1 - e^{-\lambda_e t_e}\right]}{\lambda_e Y} \quad \text{Bq/kg (wet wt.)}$$

where:

C_{vi} = resulting concentration of radionuclide i in vegetation v, Bq/kg (wet wt.)
$C_{air\ total}$ = airborne concentration of radionuclide i, Bq/m^3
E_v = fraction of the foliar deposition reaching edible portions of vegetation v (assumed equal 1 for all above-ground vegetables, and 0.1 for all below-ground vegetables), no units
F_r = fraction of the total deposition retained on plant surfaces, 0.2, no units
λ_e = effective removal constant accounting for weathering losses and radioactive decay, 1/s
t_e = effective duration of the deposition while vegetation v is growing, s
V_p = deposition velocity, m/s
Y = yield density of vegetation v, kg (wet wt.)/m^2

(4.13)

$$B_v = \frac{C_v}{C_s} \quad \text{Bq/kg veg / Bq/kg soil}$$

(4.14)

$$B_r = \frac{C_r}{C_s} \qquad \text{Bq/kg veg / Bq/kg soil.}$$

where:

B_v = soil-to-plant elemental transfer factors for vegetative portions of food crops and feed plants, kg soil (dry wt.)/kg veg (dry wt.)
B_r = soil-to-plant elemental transfer factors for non-vegetative (reproductive) portions of food crops and feed plants, kg soil (dry wt.)/kg veg (dry wt.)
C_v = elemental concentration in vegetative portions of food crops and feed plants (dry wt.) at edible maturity, Bq/kg (dry wt.)
C_r = elemental concentration in non-vegetative (reproductive) portions of food crops and feed plants (dry wt.) at edible maturity, Bq/kg (dry wt.)
C_s = elemental concentration in root zone soil, Bq/kg (dry wt.)

Note there are two classes (CSA, 1987, section 5.7) of soil-to plant transfer factors for vegetative portions of plant, B_v:

1. B_v for forage grass consumed by animals where the parameter is the ratio of the radionuclide concentration in forage (dry wt.) to that in soil
2. B_v for vegetables and fresh produce consumed by humans where the parameter is the ratio of the concentration in vegetation (fresh or wet wt.) to that in soil

The concentration of vegetation through root uptake is given by the following relationship (CSA 1987):

(4.15)

$$C_{sv} = C_s \times B_v \qquad \text{Bq/kg ….for stems and leaves}$$

(4.16)

$$C_{sv} = C_s \times B_r \qquad \text{Bq/kg …..for reproductive parts}$$

where:

C_{sv} = resulting concentration of radionuclide in vegetation, Bq/kg (dry wt.)
C_s = elemental concentration in root zone soil, Bq/kg (dry wt.)
B_v and B_r = the soil-to plant transfer factors for radionuclide *i* and vegetation type *v,* Bq/kg (wet wt.) plant per Bq/kg (dry wt.) soil

Recommended values for the soil-to-plant transfer factors for vegetative and reproductive parts of plants are given in Appendix A (Section A3.0). The values for effective soil density for surface mixing are the same as those for dry soil density; the default value is 240 kg/m^2.

Although the concentrations of radionuclides in plants are calculated in dry weight, the risk assessor may be required to convert these values to corresponding wet weight concentrations (on a plant- and -tissue specific basis) to predict ingestions exposure in a risk assessment. Default dry/wet fractions can be found in various references (e.g. Baes et al., 1984). However, it is recommended that the testing laboratory analyze the moisture content of vegetation samples at the time of analysis because default values are not precise.

4.6.3 Deposition to vegetables (irrigation)

The radionuclide concentration in vegetation due to spray irrigation by contaminated water can be determined by the following equation:

(4.17)

$$C_{v,irr} = C_{sw} \times \frac{r \times L}{\lambda_e \times Y}(1 - e^{-\lambda_e t_e}) \qquad \text{Bq/kg}$$

where:

$C_{v,irr}$ = resulting radionuclide concentration in vegetation through irrigation with contaminated water, Bq/kg (fresh wt.)
C_{sw} = radionuclide concentration in irrigation water, Bq/m^3
r = initial fraction retained on vegetation (default value of 0.05 recommended by CSA, 1987), no units
L = spray irrigation rate averaged over growing season (default value of 2.3 × 10^{-8} recommended by CSA, 1987, m^3/(s m^2)
λ_e = effective removal constant from vegetation, 1/s
Y = vegetation yield, kg/m^2 (fresh wt.)
t_e = effective duration of growing season, s

The effective removal constant from forage and crops (λ_e) is determined by radioactive decay (λ_r) and physical removal processes such as wind, rain, and plant growth (λ_p), and is given by $\lambda_e = \lambda_p + \lambda_r$. The physical removal constant is taken to be 5.73 $\times 10^{-7}$ 1/s, which corresponds to a half-life for removal of 14 days. The deposition time (t_e) is taken as 30 days for pasture grass and 50 days for all other crops. The yield densities for different types of vegetation are provided in Appendix A (Section A2.0).

4.7 Transfer to Animal Products

4.7.1 Inhalation by animals

Inhalation of air contaminated with radioactive material by animals may result in subsequent absorption and translocation to tissues. The corresponding radioactivity concentration in animal products can be estimated by the following relationship (CSA, 1987, section 5.9):

(4.18)

$$C_{inh,i} = C_{air\ total} \times I_a \times F' \qquad \text{Bq/kg}$$

where:

$C_{inh\ i}$ = resulting concentration in animal products through inhalation, Bq/kg
$C_{air\ total}$ = airborne concentration of radionuclide *i*, Bq/m^3
I_a = breathing rate of animal, m^3/d
F' = inhalation-to-animal products transfer factors indicating the fraction of the animal daily intake by inhalation and appearing in each kg of product, d/kg

An example of a method to estimate animal breathing rates was provided by Sample et al. (1997) using the allometric equations:

(4.19)

$$I_b = (0.54576\ (BW)^{0.8})/BW \qquad \text{m}^3\text{air/(d kg body wt.) (mammals)}$$

(4.20)

$$I_b = (0.40896\ (BW)^{0.77})/BW \qquad \text{m}^3\text{air/(d kg body wt.) (non-passerine birds)}$$

where:

BW = body wt., kg (live wt.)
0.54576 = constant, m^3/d
0.40896 = constant, m^3/d

Sample et al. (1997) recommended that equation (4.20) is likely also to be suitable for passerines.

Little data are available on the transfer of inhaled material to animal produce; therefore, the values of *F'* were estimated using human metabolic data (CSA, 1987). Recommended fractions of daily intake by inhalation appearing in animal produce (milk, beef, pork, eggs, and poultry meat) are provided in Appendix A (Section A3.0).

4.7.2 Ingestion by animals

The concentration in animal products, such as beef, eggs, poultry, and cows' milk from the ingestion of contaminated feed can be obtained by assessing the amount of radioactive material deposited on grasses, hay, or silage eaten by animals that become or produce animal products consumed by humans. The concentrations in animal products can be estimated by using the relationships (U.S. NRC, 1982, section 1.4; CSA, 1987, section 5.8) described in the next sections.

4.7.3 Radionuclide concentrations in meat and eggs

(4.21)

$$C_{bi} = Q_f\ F_{bi}\ (F_{pg}\ C_{pgi} + F_h\ C_{hi}) \qquad \text{Bq/kg}$$

where:

C_{bi} = resulting average concentration of radionuclide *i* in meat, Bq/kg
Q_f = feed consumption rate of the animal, kg (wet wt.)/d
F_{bi} = feed-to-meat or the feed-to-eggs transfer factors, as appropriate for radionuclide *i*, Bq/kg per Bq/d ingested
F_{pg}, F_h = fractions of the total annual feed requirement assumed to be satisfied by pasture grass, *pg*, or locally grown stored feed (hay), *h*, respectively, no units
C_{pgi} = concentration of radionuclide *i* in pasture grass, Bq/kg (wet wt.)
C_{hi} = concentration of radionuclide *i* in hay (or other stored feed), Bq/kg (wet wt.)

Recommended values for animal-specific feed consumption rates and feed-to-meat and feed-to-eggs transfer factors are given in Appendix A (Section A5.0). The fraction of total annual feed that is harvested from the contaminated area ranges (0 to 1) is site-specific, and must be determined by the risk assessor.

4.7.4 Radionuclide concentrations in milk

(4.22)

$$C_{mi} = Q_f\ F_{mi}\ (F_{pg}\ C_{pgi} + F_h\ C_{hi}) \qquad \text{Bq/L}$$

where:

C_{mi} = resulting average concentration of radionuclide *i* in milk, Bq/L
F_{mi} = feed-to-milk transfer factors for radionuclide *i*, Bq/L per Bq/d ingested
Q_f = feed consumption rate of the animal, kg (wet wt.)/d
F_{pg}, F_h = fractions of the total annual feed requirement assumed to be satisfied by pasture grass, *pg*, or locally grown stored feed (hay), *h*, respectively, no units
C_{pgi} = concentration of radionuclide *i* in pasture grass, Bq/kg (wet wt.)
C_{hi} = concentration of radionuclide *i* in hay (or other stored feed), Bq/kg (wet wt.)

Recommended values for feed consumption rate and feed-to-milk transfer factors are listed in Appendix A (Section A5.0). The fraction of total annual feed that is harvested from the contaminated area is site-specific, and ranges between 0 and 1.

4.7.5 Transfer from soil to animal products

Contaminated soil can transfer radioactivity to animals, and radionuclides can be transferred to humans through the animal products they consume. The radionuclide concentrations in animal products can be estimated by using the following equation:

(4.23)

$$C_{ai} = Q_f \; F_{si} \; F_{soil} \; C_{soil} \qquad \text{Bq/kg or Bq/L}$$

where:

C_{ai} = resulting average concentration of radionuclide *i* in animal products, Bq/kg or Bq/L for milk
Q_f = feed consumption rate of the animal, kg (wet wt.)/d
F_{si} = feed-to-meat and feed-to-milk transfer factors for radionuclide *i*, Bq/kg per Bq/ d ingested, or Bq/L per Bq/d ingested for milk
F_{soil} = fraction represented by the soil consumption rate divided by the feed consumption rate, no units
C_{soil} = concentration of radionuclide *i* in the soil, Bq/kg (dry wt.)

Recommended values for the feed-to-meat and feed-to-milk transfer factors and F_{soil} are provided in Appendix A (Section A5.0). Recommended values may be modified after the review of chemical risk assessment parameters has been completed.

4.7.6 Transfer to animal products (drinking water)

Radioactivity in drinking water provided to animals will transfer into animal products such as meat and milk. The equilibrium radionuclide concentration in animal products can be estimated for long-term exposures using the following equation (CSA 1987, section 6.4):

(4.24)

$$C_{w\,animal} = C_{sw} \; k'_w \; Q_w \; F \qquad \text{Bq/kg or Bq/L}$$

where:

$C_{w\,anima\,l}$ = resulting radionuclide concentration in animal product (milk, meat, etc.) through ingestion of water, Bq/kg or Bq/L
C_{sw} = radionuclide concentration in drinking water, Bq/L
k'_w = fraction of the annual water intake of the animal from the contaminated source, no units
Q_w = daily water intake of the animal, L/d
F = transfer factor from drinking water to animal product (milk, meat, etc.), d/L

Recommended values for the daily water intake of animals are listed in Appendix A (Section A4.0). It should be noted that daily water intake value provided in Section A4.0 must be converted before using in equation (4.24). Animal-specific transfer factors from drinking water to animal products (milk, meat, etc.) are provided as feed-to-small mammal (hare), feed-to-moose, feed-to-caribou, and feed-to-bird transfer factors in Appendix A (Section A5.0). The site-specific fraction of the animal annual water intake that arises from the contaminated source (range from 0 to 1) must be assigned by the risk assessor.

4.7.7 Transfer to fish

4.7.7.1 Fresh water environment

Under equilibrium conditions, the water-to-fish transfer factors, also known as the bioaccumulation factor, can be used to relate the amount of radionuclide in the fish through direct uptake from water, absorption from foods such as aquatic plants and other biota, and absorption from ingested sediments (CSA, 1987, section 6.6). Because the concentrations in these foods and in sediments are also related to that in water, transfer factors such as water-to-sediment are not explicitly used in determining concentration in fish. The following equation can be used to estimate concentration of a radionuclide in fish (CSA, 1987, section 6.6).

(4.26)

$$C_{wf} = C_{sw} \; F_{wf} \qquad \text{Bq/kg}$$

where:

C_{wf} = radionuclide concentration in edible fish tissues, Bq/kg
C_{sw} = radionuclide concentration in surface water, Bq/L
F_{wf} = fresh water-to-fish transfer factors, L/kg

Recommended values for fresh water-to-fish transfer factors are given in Appendix A (Section A3.0). Recommended values may be modified after the review of chemical risk assessment parameters is completed.

4.7.7.2 Marine environment

Radionuclide concentrations in edible tissues of marine fish can be calculated using the equation described above, except that the marine water-to-fish transfer factors must be used instead of the freshwater factor.

4.8 Exposure Assessment

It is impossible to anticipate all possible scenarios that may apply to all federal contaminated sites. However, the key equations applicable to screening-level radiological exposure assessment are presented in Table 4.2.

Table 4.2 Summary of Radiological Exposure Equations

Exposure Pathway	Equation	Units	Equation Number
Groundshine (semi-infinite surface)	$E_{gs} = C_{vs}\ O_s\ f_r\ (f_u + (1\text{-}f_u)\ S_g)$	Bq/m^2	4.26
Groundshine (shoreline sediments)	$E_{sh} = C_{is}\ OF_s\ W\ d_s\ DF_s$	Bq/m^2	4.27
Inhalation	$E_i = C_a\ I\ OF_i$	Bq/y	4.28
Ingestion (soil)	$E_{ing,soil} = C_{soil}\ g_{soil}\ I_{soil}$	Bq/y	4.29
Ingestion (vegetation)	$E_{ing,veg} = C_{sv}\ g_f\ I_f$	Bq/y	4.30
Ingestion (animal products)	$E_{ing,animal} = C_{a,total}\ g_f\ I_f$	Bq/y	4.31
Ingestion (water)	$E_{iw} = C_{sw}\ k'_w\ I_w$	Bq/y	4.32
Ingestion (fish)	$E_{if} = C_{wf}\ g_f\ I_f$	Bq/y	4.33
Groundshine (infinite depth)	$E_{eg} = C_{is}\ (t_{out} + f_{sh}\ t_{ind})$	Bq h/(m^3 y)	4.34
Indoor radon progeny	$E_{Rn} = C_{Ra\text{-}226}\ F_{sh}\ F_{eq}\ t_{ind} / (3{,}700 \times 170)$	WLM/y	4.35

Note: y, year; WML, working-level month.

4.8.1 Groundshine

4.8.1.1 Semi-infinite surface

Radioactivity deposited onto the ground (Section 4.5) can expose people externally while they are both indoors and outdoors. The annual exposure above a semi-infinite flat surface can be calculated based on the equations recommended in section 5.5 of the CSA standard (1987):

(4.26)

$$E_{gs} = C_{vs}\ O_s\ f_r\ (f_u + (1\text{-}f_u)\ S_g) \qquad \text{Bq/m}^2$$

where:

E_{gs} = annual exposure from radioactivity deposited on the ground, Bq/m^2
C_{vs} = radionuclide concentration deposited onto soil, Bq/m^2
O_s = fraction of annual time at site, no units
f_u = fraction of time an individual spends outdoors exposed to the ground deposit, no units (See Appendix B.)
f_r = residual dose fraction after dose reduction factor to account for non-uniformity of the ground surface (default value of 0.7 recommended by CSA, 1987), no units
S_g = shielding factor, or fraction of the out-of-door dose received indoors, reduced owing to shielding by buildings (default value of 0.4 recommended by CSA, 1987), no units

Recommended values for the fraction of time an individual spends outdoors are listed in Appendix B.

4.8.1.2 Shoreline sediments

Radioactivity deposited onto shoreline sediments (Section 4.3.2) can expose people while they are walking or fishing on the shoreline. The external exposure can be estimated using the following equation (CSA, 1987, section 6.8):

(4.27)

$$E_{sh} = C_{is}\ OF_s\ W\ d_s\ DF_s \qquad \text{Bq/m}^2$$

where:

E_{sh} = resulting external exposure from contaminated shoreline, Bq/m^2
C_{is} = radionuclide concentration in shoreline sediment, Bq/kg
OF_s = shoreline occupancy factor, fraction of annual time spent on shore, no units
W = shore-width factor that describes the shoreline exposure geometry, no units
d_s = effective density of sediment, kg/m^2
DF_s = dilution factor for shoreline deposits (≤ 1) that allows for non-equilibrium between suspended sediment and shoreline deposits, no units

Note:

1. The exposed shoreline sediment is assumed to be contaminated to some depth and at a radionuclide concentration (C_{is}) as determined in Section 4.3.2. The depth of this contaminated layer is taken to be 0.025 m; this in conjunction with a dry sediment density of 1,600 kg/m^3 gives an effective sediment density (d_s) of 40 kg/m^2. The dose contribution from radionuclides below this depth is ignored. The risk assessor must review past operations at the site and consider the need for site-specific sampling to ensure that this model is applicable.
2. The shoreline occupancy factor should be obtained from site surveys. In the absence of any commercial activity, such as clam digging, the default value for OF_s is 0.01,

which corresponds to about 100 hours per year over contaminated shoreline.
3. The value of DF_s should be conservatively taken as unity where site-specific measurements are absent. The value of DF_s may range from near unity, for bottom sediments built up from direct sedimentation from the water column, to quite small values for shorelines subject to wave action.

Recommended values for the shore width factor that describes the shoreline exposure geometry are provided in Appendix A (Section A2.0).

4.8.2 Inhalation

Airborne radioactivity at the receptor location will be inhaled and retained within the body resulting in an inhalation exposure. The inhalation exposure can be calculated based on the equation recommended in section 5.6 of the CSA Standard (1987):

(4.28)

$$E_i = C_a \; I \; OF_i \qquad \text{Bq/year}$$

where:

E_i = resulting inhalation exposure, Bq/year
C_a = radionuclide concentration in air, Bq/m^3
I = age-dependent inhalation rate, m^3/h
OF_i = occupancy factor, annual time an individual is exposed at the receptor location, h/year

Recommended values for age-dependent inhalation rates are provided in Appendix B.

4.8.3 Ingestion of soil

Radioactivity in soil (Section 4.5) that is ingested incidentally during gardening, camping, sports activities, etc., results in an ingestion exposure. The annual average exposure from soil ingestion can be calculated using the following equation:

(4.29)

$$E_{ing,soil} = C_{soil}\, g_{soil}\, I_{soil} \qquad \text{Bq/year}$$

where:

$E_{ing, soil}$ = resulting annual exposure from ingestion of soil contaminated with radioactive material, Bq/year
C_{soil} = radioactivity concentration in soil, Bq/kg (dry wt.)
I_{soil} = consumption rate of soil, kg/year (See Appendix B.)
g_{soil} = fraction of the consumed soil arising from the contaminated source (For the assessment of federal contaminated sites, g_{soil} will always be 1.)

Recommended values for the age-dependent soil ingestion rates are provided in Appendix B. The fraction consumed from a contaminated source will always be 1. Soil ingestion is not uniform throughout the day, nor is it uniform among different environments frequented throughout the day. Also, a bolus dose is unlikely, except perhaps in receptors expressing pica behaviour. However, because soil ingestion activities cannot be measured nor can they be objectively differentiated in time or space (area), it should always be assumed that 100% of a receptor's daily intake of soil arises from the contaminated site in question.

4.8.4 Ingestion of vegetation

Radioactivity in vegetables, fruits, herbs, medicinal plants (termed vegetation) (Section 4.6) that are grown in an impacted area results in an ingestion exposure when consumed. The exposure from ingestion of vegetation can be calculated using the following equation (CSA, 1987, section 5.10):

(4.30)

$$E_{ing,veg} = C_{sv}\; g_f\, I_f \qquad \text{Bq/year}$$

where:

$E_{ing, veg}$ = annual exposure from ingestion of vegetation, Bq/year
C_{sv} = radionuclide concentration in vegetation, Bq/kg (may be predicted using methods described in Section 4.6)
g_f = fraction of the consumed food from the impacted site ($0 \leq g_f \leq 1$), no units
I_f = consumption rate of vegetation, kg/year (See Appendix B.)

Recommended values for the age-dependent consumption rates of vegetables and fruits are provided in Appendix B. Site-specific values or ingestion of herbs and medicinal plants may be determined on a site-specific basis. The fraction consumed from a contaminated source is a site-specific value ranging from 0 to 1 that should be assigned by the risk assessor, and rationale should be provided in the report.

4.8.5 Ingestion of animal products

Radioactivity in animal products (Section 4.7) from animals that may frequent a site results in an ingestion exposure when consumed. The annual exposure resulting from ingestion of animal products can be calculated using the following equation (CSA, 1987, section 5.10):

(4.31)

$$E_{ing,animal} = C_{a,total}\; g_f\; I_f \qquad \text{Bq/year}$$

where:

$E_{ing,animal}$ = annual exposure resulting from ingestion of animal products, Bq/year

$C_{a,total}$ = total concentration of radionuclide *i* in animal products, Bq/kg or Bq/L (may be predicted using methods described in Section 4.7)
g_f = fraction of the consumed food arising from the impacted site ($0 \leq g_f \leq 1$), no units
I_f = consumption rate of animal products, kg/year or L/year

Recommended values for the age-dependent consumption rates of animal products are provided in Appendix B (Richardson, 1997). The fraction consumed from a contaminated source is a site-specific value ranging from 0 to 1 that should be assigned by the risk assessor, and the rationale should be provided in the report.

4.8.6 Ingestion of water

A person consuming water containing radioactivity (Section 4.3 and Section 4.4) is subject to internal exposures from retained radioactivity. Under conditions of long-term consumption, the ingestion exposure can be determined by using the following relationship (CSA, 1987, section 6.5):

(4.32)

$$E_{iw} = C_{sw}\ k''_w\ I_w \quad \text{Bq/year}$$

where:

E_{iw} = annual exposure from ingestion of radioactivity in water, Bq/year
C_{sw} = radionuclide concentration in surface water, Bq/L
k''_w = fraction of drinking water intake arising from the contaminated source, no units
I_w = annual drinking water intake, L/year

Recommended values for the age-dependent annual drinking water intakes are listed in Appendix B. The fraction of drinking water intake arising from the contaminated source is a site-specific value (0 to 1) that should be assigned by the risk assessor, and the rationale should be provided in the report.

4.8.7 Ingestion of fish

Consumption of fish containing radioactivity (Section 4.7.7) can result in internal exposure. The annual exposure from radioactivity in edible portions of fish can be calculated using the following equation (CSA, 1987, section 6.10):

(4.33)

$$E_{if} = C_{wf}\ g_f\ I_f \quad \text{Bq/year}$$

where:

E_{if} = annual exposure from ingestion of radioactivity in fish, Bq/year
C_{wf} = radionuclide concentration in edible fish tissues, Bq/kg
g_f = fraction of consumed aquatic food arising from the contaminated source, no units
I_f = annual consumption rate of fish, kg/year

It is recommended that the consumption rate of aquatic food (freshwater or marine) should be determined generally on a site-specific basis because rates can vary widely over regions of the country. Therefore, use of regional data is sometimes more applicable than using Canadian average data. Canadian average seafood intake rates are provided in Appendix B.

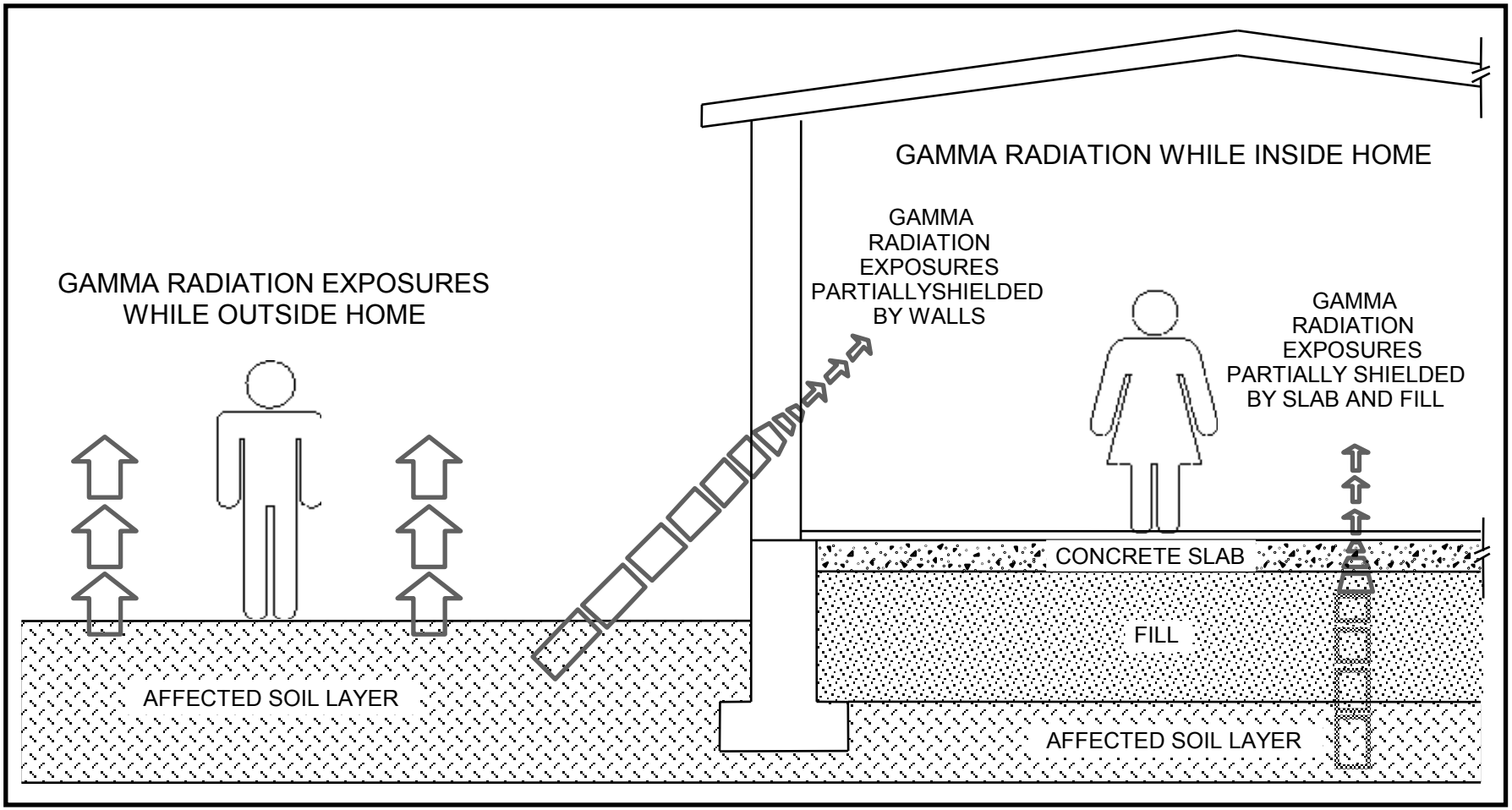
4.9 Residences on Contaminated Sites

It is very unlikely that residential housing will be built on radioactive federal contaminated sites. In the event that this occurs, this section describes the evaluation of this scenario. Residences located on contaminated sites are assumed to be built as slab-on-grade structures with no basements. Based on these building characteristics, external gamma radiation and indoor radon are the two dominant exposure pathways. Doses from other exposure pathways are considered negligible compared with the doses from the indoor radon and gamma radiation pathways. The dose from each of these pathways is controlled by the Ra-226 concentration in the soil underneath the building and in the vicinity of the residence. The following sections describe the calculation of the dose to the residents living in a house on a contaminated site from external gamma radiation and indoor radon.

4.9.1 External gamma radiation

Residents who live in houses constructed on contaminated sites will be exposed to external gamma radiation while inside and outside their homes. The external gamma radiation exposure pathways are shown in Figure 4.6. As shown in the figure, residents will be exposed to gamma radiation from soil beneath the residence and from the soil in the vicinity of the residence. However, the walls of the residence, and the slab and fill beneath the residence will provide shielding from the external gamma radiation.

Figure 4.6 Gamma Radiation Exposures to Resident of Home on Affected Soil



The annual exposure (E_{xg}) from external gamma radiation to the resident is calculated using:

(4.34)

$$E_{eg} = C_{is} \ (t_{out} + f_{sh} \ t_{ind}) \quad \text{Bq h/ (m}^3 \text{ y)}$$

where:

E_{eg} = annual exposure from external gamma radiation, Bq h/(m^3 y)
C_{is} = concentration of radionuclide *i* in soil, Bq/m^3
t_{out} = time spent outdoors, h/year
f_{sh} = shielding provided by house structure, no units (assumed = 0.2)
t_{ind} = time spent indoors, h/year

Recommended values for the time spent outdoors and indoors are provided in Appendix B. The U.S. EPA Federal Guidance Report No. 12 (U.S. EPA, 1993) provides a tabulation of gamma dose rates for soils contaminated at different depths. It is recommended that those gamma dose factors for soil contamination to an infinite depth be used in the absence of other information. When indoors, protection afforded by build-up from outdoor gamma on the ground ranges from about 0.27 for a single-storey wooden house to 0.06 for a brick house (U.S., NCRP 1984). UNSCEAR (1982) reports a worldwide average reduction factor of 0.2; this value is suggested for use in the absence of specific information.

4.9.2 Indoor radon

Residents who live in houses constructed on contaminated sites will be exposed to indoor radon levels. The relationship between the Ra-226 concentration in soil and the radon levels inside the residence is shown in Figure 4.7. As shown, the transfer of the soil gas radon level to the indoors is a combination of advective, diffusive, and ventilation processes. A schematic of this transfer process is provided in Figure 4.8. Although the process is complex and difficult to model (e.g. see UNSCEAR, 2000), a generic transfer factor of the order of 1 Bq Rn-222/m^3 per Bq Ra-226/g underlying soil is a reasonably central estimate, and is suggested for use in the absence of more specific information

The annual exposure (E_{Rn}) to the resident from radon progeny attributable to Ra-226 in soil is calculated using formula 4.35.

(4.35)

$$E_{Rn} = C_{Ra\text{-}226} \ F_{sh} \ F_{eq} \ t_{ind} / (3{,}700 \times 170) \text{ WLM/year}$$

where:

E_{Rn} = annual exposure from indoor radon, WLM/year
$C_{Ra\text{-}226}$ = Ra-226 concentration in soil, Bq/g
F_{sh} = radon soil to house transfer factor, Bq Rn-222/m^3 per Bq Ra-226/g soil
F_{eq} = indoor radon progeny equilibrium fraction, no units (assumed = 0.4)
t_{ind} = time spent indoors, h/year
3,700 = definition of working level (WL), Bq/(m^3 WL)
170 = definition of working month (WLM), h/month

Recommended values for the time spent indoors are provided in Appendix B.

Figure 4.7 Radon Exposures to Resident of Home on Affected Soil

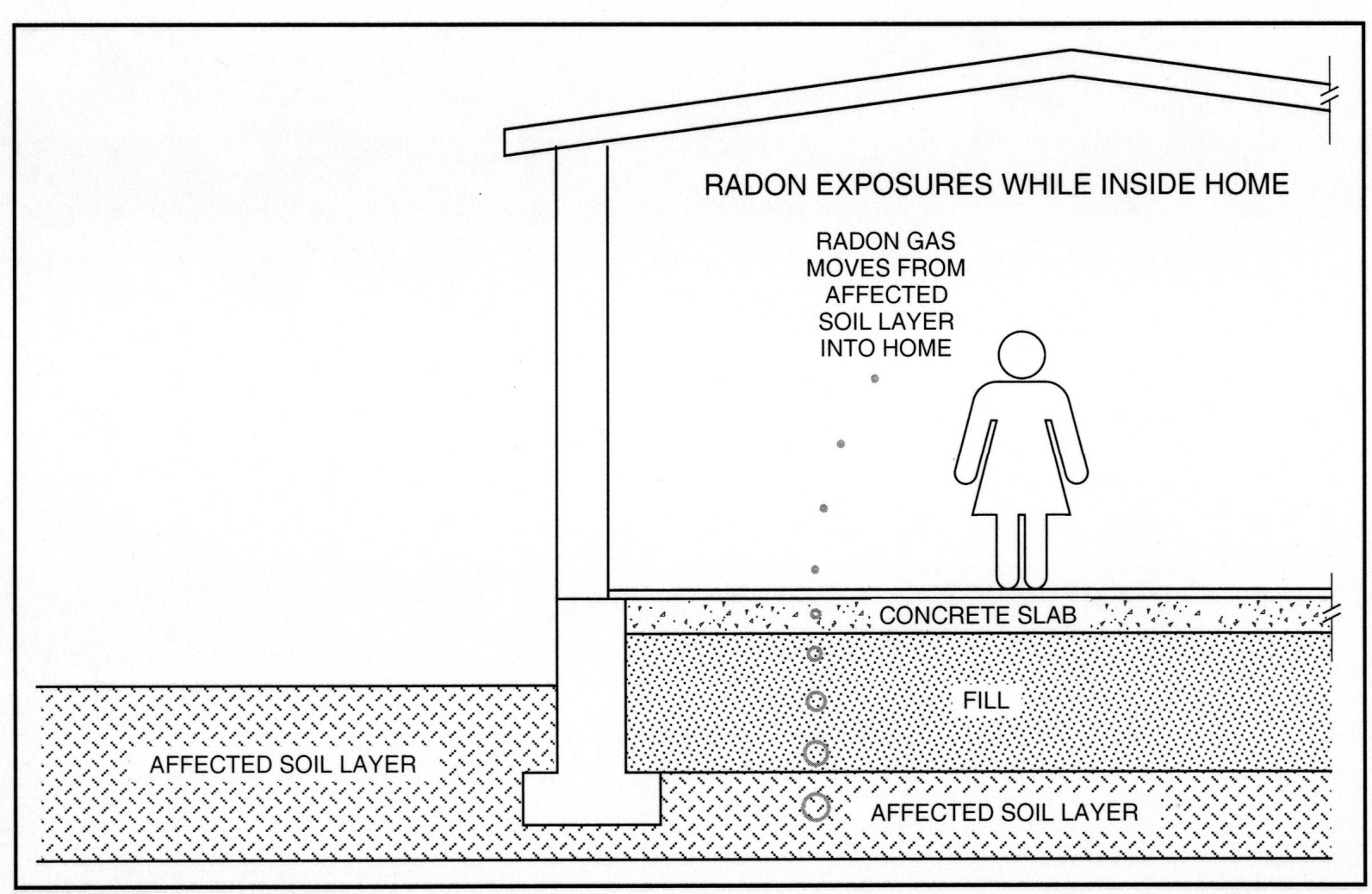
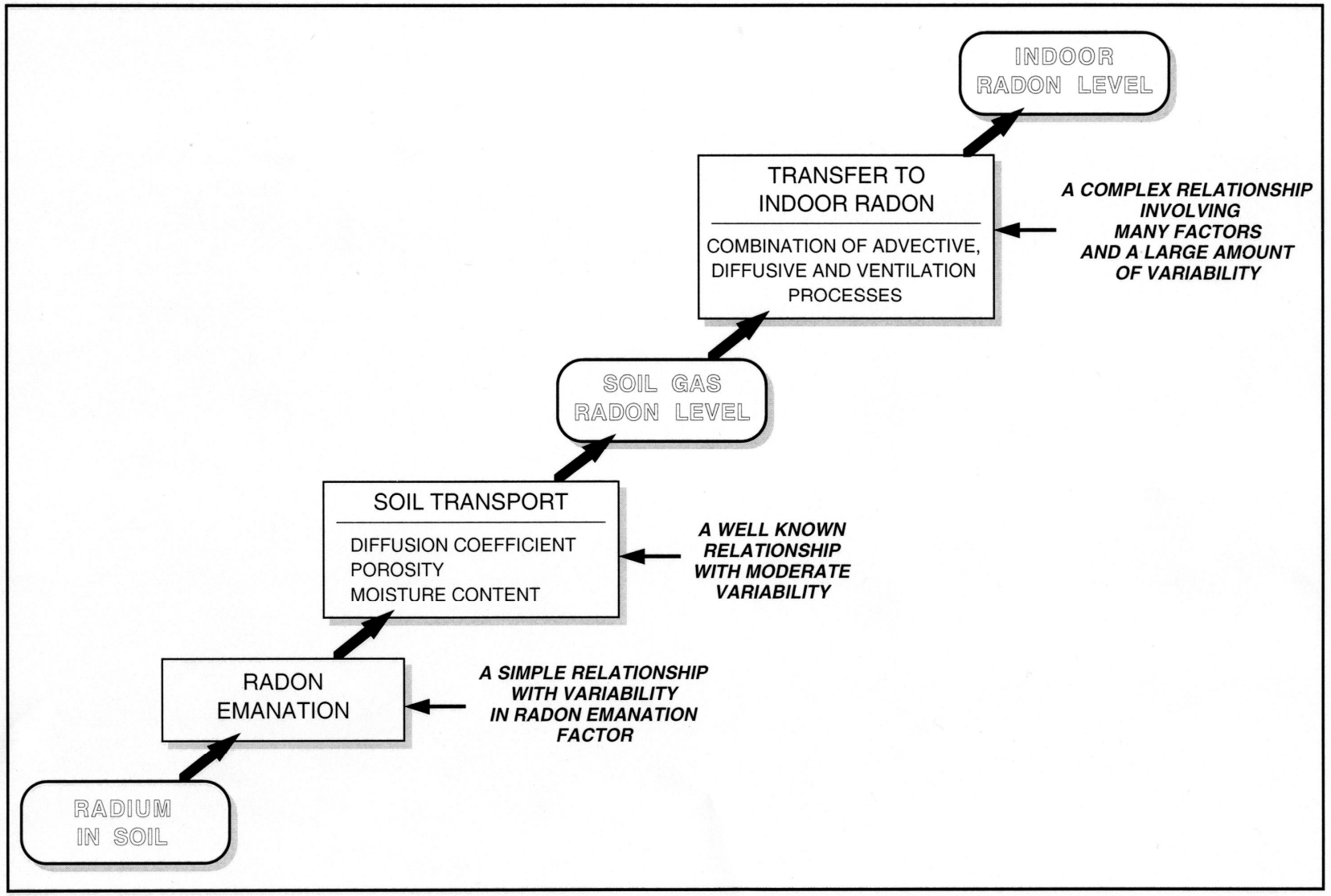


Figure 4.8 Relationship Between Radium Content in Soil and Indoor Radon Levels



4.10 Special Radionuclides – Tritium and Carbon-14

Tritium and C-14 are naturally occurring radionuclides (produced by cosmic radiation in the upper atmosphere) that are normally present in the environment in very low concentrations. Although higher-than-background concentrations are not expected at federally contaminated sites, a few comments on tritium and C-14 were considered appropriate within the scope of this manual.

The environmental transfer and exposure modelling of tritium (half-life of 12.3 years) and C-14 (half-life of 5,730 years) require special consideration because of their high mobility in the environment and the fundamental nature of hydrogen and carbon cycles in the biosphere (UNSCEAR, 2000).

4.10.1 Tritium

Tritium released to the atmosphere generally occurs as tritiated water vapour (HTO). Airborne HTO is subject to the same deposition processes as other radionuclides, but it can also diffuse into the soil pore space and directly into leaves. If the tritium concentration gradient is reversed, tritium can gradually be lost from the soil and plants to the atmosphere by evaporation and transpiration.

Tritium released to surface and groundwater also generally occurs in the HTO form, and is transported and dispersed as H_2O. Chemical and physical differences are insignificant for purposes of environmental transfer and exposure assessment.

Inhalation/skin absorption doses from tritium releases to air have been shown to be almost a factor of 10 greater than doses from corresponding ingestion pathways for milk, fruit, and vegetables (OPG, 2004). Similarly, ingestion doses (drinking water) from tritium releases to water have been shown to be a factor of 100 greater than doses from corresponding ingestion pathways for fish (OPG, 2004). Therefore, tritium exposure pathways from releases to air in this manual focus on inhalation/skin absorption, and the ingestion of garden vegetables. Similarly, tritium exposure pathways from releases to water in this manual focus on the ingestion of drinking water.

Tritium releases to the atmosphere will be dispersed by the prevailing wind, and distant concentrations in air can be calculated (see Section 4.2). Airborne tritium will exchange with water in vegetation, and CSA recommends the corresponding calculation of HTO concentration in plants using formula 4.36.

CSA (1987) reports that values of f_v range from 0.17 to 0.5, and the absolute humidity, H_a, normally ranges from 0.002 kg/m^3 in the winter months to 0.020 kg/m^3 during the spring and summer growing season. An acceptable default value for f_v / H_a is 50 m^3/kg.

4.10.1.1 Inhalation exposures

Because tritium oxide is absorbed through the skin at a rate roughly equal to the intake into the lungs, the tritium inhalation exposure calculated using the inhalation exposure equation (4.28) should be doubled when used to calculate dose described in Section 5.0 (CSA, 1987).

4.10.1.2 Ingestion exposures – garden vegetables

Tritium released to air will equilibrate in water in garden vegetables over time according to equation (4.36). Consumption of these garden vegetables will result in internal exposures from ingestion. The annual exposure from ingestion of garden vegetables can be calculated using the radioactivity concentrations in garden vegetables described in equation (4.36) and the exposure model described in equation (4.30).

4.10.1.3 Ingestion exposures – drinking water

Tritium released to surface or groundwater used as a source of drinking water will result in internal exposures to those people ingesting the water directly and indirectly through food preparation. Annual exposures from ingestion of water can be calculated using the tritium concentration in water measured or calculated using equation (4.5) and the exposure model described in equation (4.32).

(4.36)

$$C_v = C_x \ f_v / H_a \qquad \text{Bq/kg water in vegetation}$$

where:

C_v = HTO concentration in vegetation, Bq/kg water in vegetation
C_x = HTO concentration in air, Bq/m^3
f_v = ratio of specific activity of HTO in vegetation water to that in air moisture, no units
H_a = absolute humidity of air, kg/m^3

4.10.2 Carbon-14

Carbon is highly mobile and is distributed throughout the environment. C-14 released into the environment via the atmosphere enters the carbon cycle and is dispersed quickly into the atmospheric and terrestrial biosphere. The most important form of C-14 with respect to human exposure and dose is in the gaseous oxide form, CO_2, because this is the form in which carbon becomes bound in plants and ingestion; it contributes 99% of the dose from C-14. The remaining fraction of dose comes from inhalation.

Ingestion doses (milk and fruit and vegetables) from C-14 releases to air have been shown to be more than a factor of 10 greater than doses from corresponding inhalation and other pathways (OPG, 2004). Therefore, C-14 exposure pathways from releases to air focus in this manual on ingestion exposures to milk and garden vegetables.

C-14 (as CO_2) releases to the atmosphere will be dispersed by the prevailing wind, and the distant concentrations can be calculated using equation (4.1). Airborne C-14 will be incorporated into vegetation by photosynthesis. CSA recommends the calculation of C-14 in plants using the following model that assumes the specific activity of C-14 in plants is equal to the local C-14 concentration in airborne carbon:

However, this assumes that the specific activity of C-14 in carbon dioxide in air and in carbon in the vegetation is equal, and that CO_2 is present in air to the extent of 300 ppm by volume, at standard temperature and pressure. The amount of carbon in air is therefore 0.16 g/m^3.

4.10.2.1 Ingestion exposures – garden vegetables

C-14 released to air will accumulate in garden vegetables over the growing season. At equilibrium, the C-14 concentration in carbon in garden vegetables will equal the C-14 concentration in carbon in air. Consumption of these garden vegetables will result in internal exposures from ingestion. The annual exposure from ingestion of garden vegetables can be calculated using the C-14 concentrations in garden vegetables described by equation (4.37) and the exposure model described by equation 4.30.

4.10.2.2 Ingestion exposures – milk

C-14 released to air will accumulate in the pasture grass and will transfer to cows' milk over time. At equilibrium, the C-14 concentration in the carbon in the milk will equal the C-14 concentration in the carbon in the air. Consumption of the milk will result in internal exposures from ingestion. Annual exposure can be calculated using the radioactivity concentrations in cow's milk described by equation (4.37) and the exposure model described by equation (4.31).

(4.37)

$$C_v = C_x \ K_v \ / \ K_a \qquad \text{Bq/kg vegetables or cows' milk}$$

where:

C_v = C-14 concentration in vegetables or cows' milk, Bq/kg
C_x = C-14 concentration in air, Bq/m^3
K_v = carbon concentration in fresh vegetables (60) or cows' milk (440), gC/kg (fresh wt. vegetables and dry wt. forage)
K_a = carbon concentration in air (0.16), g/m^3

5.0 RADIATION DOSE ASSESSMENT

In radiological risk assessments, sources of radiation, such as gamma radiation, that are external to the body (there is no analogue in chemical risk assessments) and internal sources of ionizing radiation need to be converted to a common metric **effective dose** (measured in Sv), and then summed. The total risk from the combined exposures is proportional to the effective dose.

Effective dose considers the individual dose to each organ and the relative susceptibility (i.e. cancer risk) of each organ to that dose. The organ doses include those doses received at the time of exposure, as well as the committed dose that arises because of the ongoing irradiation caused by internally deposited radionuclides. As previously noted in Section 2.0, the absorbed dose is multiplied by a radiation weighting factor to give a (risk) equivalent dose reported in Sv. Not all organs and tissues have the same sensitivity to ionizing radiation, therefore the ICRP (ICRP, 1991) has developed tissue weighting factors. The effective dose, also reported in Sv, is calculated by multiplying the effective dose to each tissue/organ by the appropriate tissue/organ weighting factor, and then summing all the tissue/organ doses.

This section discusses the various factors that affect the calculation of dose, given the radionuclide concentrations in various environmental media such as soil, water, or fish (see Section 4.0).

5.1 Receptor Characterization

To convert concentration of radionuclides in the various environmental media to dose, it is necessary to consider the characteristics of possible receptors, including for example, the inhalation rate, water ingestion rate, and food consumption pattern.

Both recommended deterministic values and default probabilistic distributions of receptor characteristics to be used in radiological risk assessments are provided in Appendix B. Receptor characteristics discussed in Appendix B include body weight, soil/dust ingestion rate, inhalation rate, water ingestion rate, time spent indoors and outdoors, skin surface area, soil adherence factors, human milk intake rate, the fraction of produce acquired from backyard gardens, and food consumption rates. At the conceptual model stage, the exposure pathways, and hence the characteristics that are needed should have been determined.

The characteristics provided in Appendix B are developed from the receptor characteristics typically used in chemical risk assessments in Canada (e.g. HC, 1994; Richardson, 1997; Leech et al., 2002; MOE, 2002; HC, 2003). Thus, this step provides some harmonization between chemical and radiological risk assessments.

In radiological risk assessments, the resultant radiation dose depends on the age-dependent metabolic characteristics of the receptor (e.g. mass, size, breathing rate). To account for these differences, the ICRP provides DCFs and metabolic factors for specific age groups that are considered to cover a range of ages, as shown in Table 5.1.

The age classes available for DCFs discussed in the preceding table differ from the receptor characteristics provided in Appendix B; thus, some assumptions have been made so that the same receptor characteristics can be used for both radiological and chemical risks. The ICRP 3-month-old infant (0–12 months) has been assumed to be adequately represented by the North American 0–6-month infant, whereas the ICRP 1-year-old toddler (1–2 years) was assumed to be represented by the North American 0.5–4 year-old toddler. The ICRP 5-year-old (2–7 years) and 10-year-old (7–12 years) children were assumed to be represented by the North American 5–11-year-old child,

Table 5.1 Age Classes for Dose Conversion Factors

ICRP age	Age range covered	Recommended range for FCSAP risk assessments
3 months	infant 0–12 months	infant 0–6 months
1 year	1–2 years	toddler 0.5–4 years
5 years	more than 2– 7 years	child 5–11 years
10 years	more than 7–12 years	child 5–11 years
15 years	more than 12–17 years	teen 12–19 years
adult	more than 17 years	adult 20+

Source: ICRP, 1996.

whereas the ICRP 15-year-old (12–17 years) teen was assumed to be represented by the North American 12–19-year-old teen. The ICRP adult (>17 years) was assumed to be represented by the North American adult (20 years +).

Dietary characteristics for typical Canadians are also provided in Appendix B; however, it should be recognized that First Nation people who may have access to radiological contaminated sites may have very different dietary habits because they may rely heavily on traditional sources of food. Depending on the region of the country they are from, the dietary characteristics can be quite different. There are several different dietary surveys for First Nation people in the Northwest Territories (Receveur et al., 1996, 1998), Inuvialuit (Kuhnlein et al., 2000), and northern Saskatchewan (Can North, 2000) that should be consulted to obtain the appropriate dietary characteristics for First Nation people if they are considered in the risk assessment. Golder Associates Ltd. (2005) has also prepared a document on country food consumption. Other site-specific references may be applicable on a site-specific basis. In addition, the *Compendium of Canadian Human Exposure Factors for Risk Assessment* (Richardson, 1997) has some general information on First Nation dietary characteristics. However, numerous dietary surveys have been performed (see Appendix D) that should be consulted if First Nation or Inuit communities represent a receptor group.

In risk assessment, "critical receptors" may be defined in a number of ways, such as:

- a community or population group that will experience relatively high exposure because of proximity to a site and/or increased frequency, duration, and/or intensity of interaction with the site;
- toddlers who receive the greatest overall dose because of higher intakes (via ingestion, inhalation, etc.) per unit body weight than all other age groups; and
- a sensitive life stage, or a genetic predisposition to effects, that will result in that receptor being the most likely (sensitive) to suffer effects because of exposure.

Risk assessments must identify the critical receptor group(s) that apply to the site in question. For radiological protection, the critical group is the group(s) thought most likely to receive the largest exposures for a particular site and scenario (ICRP, 1991). It should be noted that screening-level risk assessments should evaluate the critical receptor group(s), and should also incorporate conservative assumptions to define reasonable worst case or reasonable maximum exposure. Complex site-specific risk assessments must also include other receptor groups and focus on average or typical exposure, but the critical receptor group(s) must still be addressed.

Screening radiological risk assessments are often done for adult receptors, but doses can be estimated for other age groups as needed.

5.2 Overview of Dose Models

As discussed previously in Section 2.0, in radiological risk assessment, the potential doses from all relevant radionuclides are summed to provide an estimate of the total (effective) dose that is then compared with the dose limit or other relevant dose objectives.

To convert internal sources of radiation resulting from the ingestion or inhalation of radionuclides to dose, risk assessors make use of standard DCFs.

The ICRP publishes DCFs for both workers in its Publication 68 (ICRP, 1994a) and members of the public in its Publication 72 (ICRP, 1996)) for essentially all known radionuclides that could be encountered in occupational or environmental settings. The ICRP DCFs are accepted for use in risk assessments by regulatory and other agencies around the world.

The DCFs take into account the specific radiological/metabolic characteristics of each radionuclide by calculating the dose resulting from unit intakes of the radionuclide, measured in Bq. The DCFs are given in units of Sv/Bq. Biokinetic/pharmokinetic models, mostly based on data obtained from research with non-radioactive chemicals/elements, are used to follow the time-dependent transfer of the radionuclide through the body, resulting either in deposition in various body organs and/or elimination via feces and urine. The models account for radioactive decay and the ongoing dose received after intakes cease, and are specific to each element. The models account for, among many other factors, the gut-to-blood transfer (solubility) of the radionuclide in the body (dependent on the chemical form of the radionuclide). The age-dependency of all these factors, including body organ masses, inhalation rates, etc., is also taken into account.

It is recommended that ICRP dose conversion factors for radionuclides be used in the radiological risk assessment to convert intake via inhalation or ingestion to dose.

For the specific case of pregnant women, the ICRP policy is that the fetus should be provided a level of radiation protection that is broadly comparable with that provided for members of the public. Therefore, the ICRP also provides dose coefficients for the embryo and fetus after intakes of radionuclides by the mother (ICRP, 2001, 2003). A CD-ROM (ICRP, 2002a) is also available; it further adds to Publication 88 (ICRP, 2001), covering various organs and tissues of the offspring up to birth and to various times after birth (ICRP, 2003). Except for unlikely circumstances with the potential for

elevated internal exposures, the dose to the fetus need not be calculated for screening-level or complex risk assessments.

5.3 Inhalation Models

Inhalation dose conversion factors that should be used for the uranium series of radionuclides are provided in Table 5.1.

To calculate inhalation DCFs, the ICRP respiratory tract model was used. The model, referred to as the human respiratory tract model, was adopted by the ICRP in 1994, and it is described in the ICRP Publication 66 (ICRP, 1994b). Guidance on the use of the model is also provided in the ICRP Supporting Guidance 3 (ICRP, 2002b). The model makes use of a wide range of parameter values, such as particle size, lung clearance rates, and breathing rates. In general, the risk assessor should use default or reference values for DCFs that have been selected by the ICRP to be typical representative values in its Publication 72 (ICRP, 1996). This reference contains the default DCFs recommended by the ICRP for environmental assessments. Examples of DCF values for inhalation that should be used in a typical risk assessment are shown in Table 5.2. As well, the ICRP CD-ROM (ICRP, 2002a) is a database of DCFs for both workers and members of the public. It provides ingestion and inhalation DCFs for a wider range of parameter values (e.g. varying particle sizes from 0.001 to 10 µm), as well as doses for specific body organs for a range of integration times. The ICRP Supporting Guidance 3 (ICRP, 2002b) provides guidance on when to use different parameter values for specific situations where the default values are inappropriate.

The DCF dependencies on various significant parameters are discussed below.

5.3.1 Particle size

Particle size is a very important parameter for determining how much and where inhaled materials are deposited in the respiratory tract. The ICRP measures particle size in terms of the activity median aerodynamic diameter (AMAD). Fifty percent of the activity of the particle is greater than the AMAD. The AMAD represents the proportion of a particle cloud in a particular size range – not the average or maximum aerodynamic diameter of particles as typically used in the measurement and assessment of airborne particulate matter. The ICRP Publication 66 (ICRP, 1994b) on the respiratory tract model provides deposition values for various regions of the respiratory tract for aerosols ranging from 0.0006 to 20 µm AMAD. Values are given for workers (both normal nose breathers and habitual mouth breathers) and for members of The ICRP Publication 66 (ICRP, 1994b) on the respiratory tract model provides deposition values for various regions of the respiratory tract for aerosols ranging from 0.0006 to 20 µm AMAD. Values are given for workers (both normal nose breathers and habitual mouth breathers) and for members of the public (normal nose breathers) for each ICRP age group, and at four reference levels of activity: sleeping, sitting, light exercise, and heavy exercise. Inhalation DCFs for have been calculated, and are available for 10 aerosol sizes (0.001, 0.003, 0.01, 0.03, 0.1, 0.3, 1, 3, 5, and 10 µm AMAD (ICRP, 2002a). The appropriate DCF should be selected based on the aerosol size. An example of the effect of particle size on the DCF(Sv per Bq inhaled) is provided for U-238 in Table 5.3.

Table 5.2 Inhalation Dose Conversion Factors for Members of the Public

Radionuclide	Type*	Adult†	Child‡
		DCF (Sv/Bq)	DCF (Sv/Bq)
Uranium-238	S	8.0×10^{-6}	1.6×10^{-5}
Uranium-234	S	9.4×10^{-6}	1.9×10^{-5}
Uranium-nat§	S	8.7×10^{-6}	1.8×10^{-5}
Thorium-230	S	1.4×10^{-5}	2.4×10^{-5}
Radium-226	S	9.5×10^{-6}	1.9×10^{-5}
Lead-210	S	5.6×10^{-6}	1.1×10^{-5}
Polonium-210	S	4.3×10^{-6}	8.6×10^{-6}

Note: DCF, dose conversion factor.

*Type S materials – slow rate at which radionuclides are cleared from the lungs.

†1µm activity median aerodynamic diameter particles.

‡Values for a 5-year-old child.

§The calculated value for natural uranium (uranium-nat) assumes equal activities of U-238 and U-234, and that the small (<5%) activity contribution from U-235 can be ignored; 1 Bq of uranium-nat is assumed to contain 0.5 Bq each of U-238 and U-234.

Source: ICRP, 1996.

Table 5.3 Dependence on Particle Size of Uranium-238 Inhalation Dose Conversion Factors for Adults

Size (μm)	0.03	0.1	0.3	1	3	5	10
DCF (Sv/Bq)	4.0×10^{-5}	2.1×10^{-5}	1.1×10^{-5}	8.0×10^{-6}	8.1×10^{-6}	6.5×10^{-6}	4.0×10^{-6}

Note: DCF, dose conversion factor.
Source: ICRP, 1996.

In the absence of site-specific information and consistent with the recommendation of the ICRP, a particle size (AMAD) of 1 μm should be assumed as a default value for members of the public[7] (ICRP, 1996).

5.3.2 Solubility class

The solubility of the particle in the respiratory tract is an important parameter impacting the rate of clearance of the particle from the respiratory tract to other parts of the body. The rate of clearance of the particles is dependent on the chemical form of the inhaled material and classified by the ICRP as follows:

Type F materials – deposited materials readily absorbed into the blood from the respiratory tract (fast rate of solubilization)

Type M materials – deposited materials with intermediate rates of absorption into the blood from the respiratory tract (moderate rate of solubilization)

Type S materials – deposited materials relatively insoluble in the respiratory tract (slow rate of solubilization)

The ICRP recommended inhalation types for various elements are provided in Table 5.4.

[7] For workers who are generally closer to dust-generating activities, the default particle size of 5 μm (ICRP, 1994) is often used. The risk assessor will need to make judgments on a case-by-case basis concerning this factor.

Table 5.4 Lung Clearance Types for Various Radionuclides and Compounds for Members of the Public

Radionuclide	Type	Compounds
Uranium	F	most tetravalent compounds, e.g. UF_6, UO_2F_2, $UO_2(NO_3)_2$
	M	less soluble compounds, e.g. UO_3, UF_4, UCl_4 and most other hexavalent compounds
	S	highly insoluble compounds, e.g. UO_2 and U_3O_8
Thorium	S	oxides and hydroxides
	M	unspecified compounds
Radium	M	all compounds
Lead	F	all compounds

Source: ICRP, 1996.

The dependence of the inhalation DCF on particle solubility, illustrated in Table 5.5, is for U-238 for adult members of the public.

Table 5.5 Variation of Uranium-238 Dose Conversion Factors with Inhalation Type for Adults

Type	F	M	S
DCF (Sv/Bq)	5.0×10^{-7}	2.9×10^{-6}	8.0×10^{-6}

Note: DCF, dose conversion factor.

Source: ICRP, 1996.

For the case of U-238, the variation in the DCF with inhalation type is more than a factor of 10. The S type particles result in the largest DCF; this is the case for most radionuclides.

In the absence of specific information for the radionuclide of interest, the more restrictive of the ICRP DCFs should be used. This is generally the F type particle.

5.3.3 Age of receptor

The inhalation DCFs are necessarily dependent on the age-dependent metabolic parameters of the receptors. As previously noted, the ICRP respiratory tract model accounts for these factors. Table 5.6 shows the age dependence of the inhalation DCF for U-238 for default particle size (1 µm) for members of the public.

Table 5.6 Age-Dependence for U-238 Inhalation Dose Conversion Factors for Members of the Public

Age	3 months	1 year	5 years	10 years	15 years	Adult
DCF (Sv/Bq)	2.9×10^{-5}	2.5×10^{-5}	1.6×10^{-5}	1.0×10^{-5}	8.7×10^{-6}	8.0×10^{-6}

Note: Table is based on a 1 µm activity median aerodynamic diameter. DFC, dose conversion factor.

Source: ICRP, 1996.

The inhalation DCFs decrease with age. For the case of U-238, the variation with age is approximately a factor of four reductions from infants to adults.

5.3.4 Effective dose versus organ dose

The ICRP inhalation DCFs are generally provided in terms of effective dose per amount of radioactivity inhaled (Sv/Bq). However, to calculate the effective dose, the dose to each body organ must first be calculated. Therefore, organ doses, unweighted for various sensitivities to radiation dose, are also available from the ICRP respiratory tract model. This can be quite useful (e.g. when both chemical and radiological exposures and the combined impacts to specific organs are being considered) because the ICRP model provides estimates of the amount of a radionuclide in body organs. The model also provides the dose commitment to these organs as a function of time after intake. An example of this output for U-238 is provided in Table 5.7.

Table 5.7 Organ Doses for Inhalation of Uranium-238 for Adults

	Organ doses (Sv/Bq)				
Time after intake	7 days	30 days	1 year	10 years	50 years
Bladder wall	1.10×10^{-11}	3.30×10^{-11}	2.40×10^{-10}	2.60×10^{-9}	1.50×10^{-8}
Bone surface	1.20×10^{-9}	3.40×10^{-9}	2.50×10^{-8}	1.90×10^{-7}	4.60×10^{-7}
Breast	8.80×10^{-12}	3.50×10^{-11}	3.20×10^{-10}	2.90×10^{-9}	1.50×10^{-8}
Esophagus	9.00×10^{-12}	3.60×10^{-11}	3.40×10^{-10}	3.00×10^{-9}	1.50×10^{-8}
Stomach wall	2.20×10^{-10}	2.70×10^{-10}	5.90×10^{-10}	3.10×10^{-9}	1.50×10^{-8}
Colon	6.00×10^{-9}	6.80×10^{-9}	9.30×10^{-9}	1.40×10^{-8}	2.60×10^{-8}
Liver	3.90×10^{-11}	1.20×10^{-10}	1.20×10^{-9}	2.00×10^{-8}	6.20×10^{-8}
Ovaries	9.90×10^{-12}	3.10×10^{-11}	2.40×10^{-10}	2.60×10^{-9}	1.50×10^{-8}
Red marrow	1.20×10^{-10}	3.40×10^{-10}	2.80×10^{-9}	2.20×10^{-8}	4.90×10^{-8}
Lungs	4.50×10^{-6}	1.40×10^{-5}	3.20×10^{-5}	5.70×10^{-5}	6.70×10^{-5}
Skin	8.00×10^{-12}	3.00×10^{-11}	2.40×10^{-10}	2.60×10^{-9}	1.50×10^{-8}
Testes	7.90×10^{-12}	2.90×10^{-11}	2.40×10^{-10}	2.60×10^{-9}	1.50×10^{-8}
Thyroid	8.20×10^{-12}	3.10×10^{-11}	2.60×10^{-10}	2.70×10^{-9}	1.50×10^{-8}
Remainder	1.50×10^{-10}	5.80×10^{-10}	5.50×10^{-9}	1.80×10^{-8}	3.10×10^{-8}
Effective dose	5.40×10^{-7}	1.70×10^{-6}	3.90×10^{-6}	6.90×10^{-6}	8.00×10^{-6}

Note: Table is based on Type S materials, 1 µm activity median aerodynamic. 1 µm is the default value for public exposures assumed in absence of specific data.

In this case, the lungs receive the highest dose. For intakes of more soluble U-238 (Type F), the bone surface, followed by the liver, would be the most exposed organs. This is because the Type F U-238 would be cleared from the lungs before the lungs receive significant doses.

It should be noted, for typical radiological assessments carried out under the FCSAP program, the risk assessor need not consider organ dose. Additional information is provided in Section 5.4.3.

5.4 Ingestion Models

As previously noted, the ICRP uses biokinetic/pharmokinetic models to convert intakes of radioactivity via ingestion to radiation doses. However, unlike the ICRP respiratory tract model that is applied to all radionuclides (with radionuclide-specific parameter values), the ICRP has developed ingestion dose models specific to each radionuclide. Radionuclide specific factors such as uptake to blood (solubility), and inter-organ transfer rates, distribution, and retention are considered. The age-dependency of these factors, as well as that of organ masses, is also taken into account.

For uranium, the bone, liver, and kidneys are the principal organs of interest. The nephrotoxicity of uranium due to its chemical properties is generally more significant than its radiotoxicity. Ingestion DCFs that should be used for the uranium series radionuclides are provided in Table 5.7: those recommended for use in radiological risk assessments are provided in Table 5.8.

Table 5.8 Ingestion Dose Conversion Factors for Uranium Series Radionuclides for Members of the Public

Radionuclide	f_1*	Adult†	Child‡
		DCF (Sv/Bq)	DCF (Sv/Bq)
Uranium-238	0.02	4.5×10^{-8}	8.0×10^{-8}
Uranium-234	0.02	4.9×10^{-8}	8.8×10^{-8}
Uranium-nat§		4.7×10^{-8}	8.4×10^{-8}
Thorium-230	5.0×10^{-4}	2.1×10^{-7}	3.1×10^{-7}
Radium-226	0.2	2.8×10^{-7}	6.2×10^{-7}
Lead-210	0.2	6.9×10^{-7}	2.2×10^{-6}
Polonium-210	0.5	1.2×10^{-6}	4.4×10^{-6}

*Gut-to-blood transfer factor.

†Default values recommended for adult members of the public.

‡Values for a 5-year-old child.

§The calculated value for natural uranium (uranium-nat) assumes equal activities of U-238 and U-234, and that the small (<5%) activity contribution from U-235 can be ignored; 1 Bq of uranium-nat contains 0.5 Bq each of U-238 and U-234.

Source: ICRP, 1996.

5.4.1 Solubility class

The transfer of ingested radioactive material throughout the body and the resultant radiation dose is strongly dependent on the solubility of ingested material. The ICRP parameterizes the solubility in its models by means of the gut-to-blood transfer factor (f_1). Based on studies on dietary uptake, animal studies, and in vitro laboratory studies, the ICRP recommends default solubility or f_1 factors for all the elements for which they provide DCFs. Examples of the recommended f_1 values for various elements and two age groups are provided in Table 5.9. The different f_1 values reflect the effect of greatly differing metabolic behaviour (see Section 5.4.2).

Table 5.9 Default f_1 Values for Various Radionuclides and Compounds for Members of the Public

Radionuclide	f_1*	Age (years)
Uranium	0.04	< 1
	0.02	> 1
Thorium	0.005	< 1
	0.0005	> 1
Radium	0.4	< 1
	0.2	> 1
Lead	0.6	< 1
	0.2	> 1
Polonium	1.0	< 1
	0.5	> 1
	0.5	> 1

In the absence of specific information for the radionuclide of interest, the ICRP-recommended default f_1 values should be used, as provided in Table 5.8.

* Gut-to-blood transfer factor.
Source: ICRP, 1996

Studies by Health Canada (Limson Zamora et al., 2002) on the uptake of uranium from drinking water and food have supported the ICRP f1 uranium values. The ICRP f1 values for all elements may be found in the ICRP Publication 72 (ICRP, 1996).

5.4.2 Receptor age

Similar to the DCFs for inhalation, the DCFs for ingestion are dependent on metabolic characteristics of the receptor and therefore vary with age. Table 5.10 shows the age-dependence of ingestion DCFs for members of the public.

Table 5.10 Age-Dependence for Uranium-238 Ingestion Dose Conversion Factors for Members of the Public

Age	3 months	1 year	5 years	10 years	15years	Adult
DCF (Sv/Bq)	3.4×10^{-7}	1.2×10^{-7}	8.0×10^{-8}	6.8×10^{-8}	6.7×10^{-8}	4.5×10^{-8}

Note: DCF, dose conversion factor.
Source: ICRP, 1996.

As with inhalation DCFs, ingestion DCFs decrease with age. For the case of U-238, the variation with age is by a factor of 8 from 3-month-old infants to adults.

5.4.3 Effective dose versus organ dose

The ICRP concept of effective dose should be used unless there are compelling reasons otherwise.

Similar to inhalation DCFs, compilations of the ICRP ingestion DCFs are generally in terms of effective dose per amount of radioactivity inhaled (Sv/Bq). It is the effective dose (and associated risk) that is the most significant metric in radiological risk assessments. This is because environmental exposures do not lead to radiation doses to individual organs that exceed threshold dose levels (e.g. levels that are typically considered safe). However, organ doses, unweighted for their various sensitivities to radiation dose, are also available from the ICRP models. The models also provide the dose commitment to these organs as a function of time after intake. An example of this output for U-238 is provided in Table 5.11.

Table 5.11 Organ Doses from Ingestion of Uranium-238 in Food or Water for Adults

Time after intake	Organ doses (Sv/Bq)				
	7 days	30 days	1 year	10 years	50 years
Adrenals	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.5×10^{-8}
Bladder wall	2.8×10^{-10}	7.0×10^{-10}	1.6×10^{-9}	6.4×10^{-9}	2.5×10^{-8}
Bone surface	3.4×10^{-8}	7.2×10^{-8}	1.7×10^{-7}	4.2×10^{-7}	7.1×10^{-7}
Brain	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Breast	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Esophagus	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Stomach wall	1.2×10^{-9}	1.6×10^{-9}	2.5×10^{-9}	7.3×10^{-9}	2.5×10^{-8}
Small intestine wall	2.6×10^{-9}	3.0×10^{-9}	3.9×10^{-9}	8.7×10^{-9}	2.7×10^{-8}
Upper large intestine wall	1.5×10^{-8}	1.5×10^{-8}	1.6×10^{-8}	2.1×10^{-8}	3.9×10^{-8}
Lower large intestine wall	4.5×10^{-8}	4.5×10^{-8}	4.6×10^{-8}	5.1×10^{-8}	6.9×10^{-8}
Colon	2.8×10^{-8}	2.8×10^{-8}	2.9×10^{-8}	3.4×10^{-8}	5.2×10^{-8}
Kidneys	4.7×10^{-8}	1.0×10^{-7}	1.3×10^{-7}	2.2×10^{-7}	2.5×10^{-7}
Liver	1.1×10^{-9}	2.5×10^{-9}	8.5×10^{-9}	5.0×10^{-8}	9.6×10^{-8}
Muscle	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Ovaries	2.2×10^{-10}	6.4×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.5×10^{-8}
Pancreas	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Red marrow	3.3×10^{-9}	7.2×10^{9}	1.8×10^{-8}	4.9×10^{-8}	7.5×10^{-8}
Endotracheal airways	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Lungs	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.5×10^{-8}
Skin	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Spleen	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Testes	2.2×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.5×10^{-8}
Ovaries	2.2×10^{-10}	6.4×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.5×10^{-8}
Thymus	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Thyroid	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Uterus	2.2×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}
Remainder	7.4×10^{-10}	1.7×10^{-9}	2.9×10^{-9}	8.5×10^{-9}	2.7×10^{-8}
Effective dose	4.4×10^{-9}	5.7×10^{-9}	9.1×10^{-9}	2.1×10^{-8}	4.5×10^{-8}
Thymus	2.1×10^{-10}	6.3×10^{-10}	1.5×10^{-9}	6.3×10^{-9}	2.4×10^{-8}

Source: ICRP, 2002a.

In the case of ingestion of U-238, the organs that receive the highest radiation dose are the bone surface, followed by the kidney. It is interesting to note that 1 year after intake, about 20% of the lifetime (50 years) effective dose has already been received. This is a consequence of the ongoing elimination of uranium from the body.

As discussed previously, risk assessments under the FCSAP program need not consider organ dose.

5.4.3.1 Kidney burden of uranium

The ICRP models can also be used to provide organ burdens in mass rather than radiation dose units. This is important when chemical toxicity is an issue, such as for uranium intakes. The risk assessor needs to consider on a case-by-case basis whether or not the chemical toxicity of uranium needs to be carried out in addition to its radioactivity. For the specific case of uranium intakes, Chen et al. (2004) have used the most recent ICRP models to generate a compilation of kidney burdens (the most sensitive organ to uranium intakes) from common intakes. Calculations were made for four age groups from infant to adult. For all age groups, the modelling indicates that chronic long-term ingestion will result in a kidney burden of about 6.6% of the daily uranium intake.

5.5 Tritium

The ICRP Publication 72 (ICRP, 1996) provides ingestion and inhalation DCFs for tritium. Solubility fractions of $f_1 = 1.0$ are assumed for tritiated water (HTO) and for organically bound tritium, such as may be found in ingested food. However, unlike other radionuclides, tritium may also expose the body via dermal uptake, such as when surrounded by an atmosphere containing tritium (primarily as HTO). Therefore, to allow for HTO uptakes via skin absorption, the inhalation DCF for tritium should be multiplied by a factor of 2 when calculating tritium doses via the dermal pathway (CSA, 1987).

5.6 Dose from External Exposures

The dose from external sources of radiation is handled differently than internal exposures. The dose rate depends on the radionuclide, the location and dimensions of the source (point, planar, etc.), the geometry of the irradiation (from the front, from the ground, isotropic, etc.), and the size (age) of the irradiated person. Although not provided by the ICRP for many situations, radionuclide-specific external DCFs for specific irradiation geometries (ground-based sources of varying thicknesses, and air and water-based sources) are available from the U.S. EPA Federal Guidance Report No. 12 (U.S. EPA, 1993) and updates, or from the UNSCEAR (2000) report for the natural uranium, thorium, and potassium radionuclides.

In general, the external gamma rate is calculated as discussed in Section 4.0. Gamma radiation exposure rates at environmental levels are often measured and calculated in units of microroentgen per hour (μR/h). To convert exposures to radiation doses, the following conversion factor is used:

(5.1)

$$1\ \mu R = 1 \times 10^{-5}\ mSv$$

It should be noted that this is a conservative DCF (i.e. tends to overestimate the dose) for all ages when used in typical environmental situations. For example, based on the UNSCEAR (2000) report on ionizing radiation, the DCF for gamma radiation from uranium or thorium series radionuclides in the ground ranges from about 0.78×10^{-5} mSv/μR (infants) to 0.61×10^{-5} mSv/μR (adults).

This gamma exposure rate (μSv/h) should be multiplied by the time spent on site to determine the exposure to external radiation. For more specific exposure scenarios, the risk assessor can calculate external dose using commercially available models, such as MicroShield™ (Framatome ANP Inc., 2004).

5.7 Dose from Radon and Thoron

Rn-222, the radioactive decay product of Ra-226, contributes over 50% of the dose from background radiation. The calculation of dose from exposure to radon requires special considerations.

Being an inert gas, radon itself produces very little dose in an exposed person. It is the short-lived radioactive decay products of radon (RnD) that attach to particles and to the respiratory tract when inhaled that produce the dose. Conventionally, however, the RnD dose is attributed to the radon. The RnD (Po-218, Pb-214, Bi-214) are solid elements, with radioactive half-lives in the order of 3 minutes to 30 minutes. When radon is emitted from radium-bearing materials, the solid decay products are not emitted. The RnD "grow" in time toward equilibrium with the airborne radon. For a given concentration of radon, the dose is dependent on degree of equilibrium between the radon and the RnD.

In enclosed spaces, such as indoors, the concentrations of the radon and RnD build up to levels determined primarily by the ingress rate of radon and the ventilation rate of the structure. Under typical conditions, the equilibrium fraction indoors is assumed to be 0.4 (UNSCEAR, 2000; ICRP, 1993). (See also Section 4.0.)

Outdoors, the radon equilibrium fraction is a function of the travel time from the radon source (distance divided by wind speed), as well as the rate of deposition of decay products attached to particles. The outdoor radon equilibrium fraction can be obtained from Figure 5.1. The shorter the distance, the smaller the ingrowth time and corresponding dose for the same radon concentration (see Figure 5.1). Relative to contaminated sites, taking account of the times spent indoors and outdoors, and the low equilibrium fractions in the general vicinity of the sites, the dose from radon outdoors is a small fraction (<5%) of the dose indoors, and is not quantified in screening-level risk assessments.

Exposure to radon is often measured in historical units. Exposure for 170 hours (a working month) to 3,700 Bq/m^3 of radon in equilibrium with its decay products (1 WL) results in an exposure of 1 WLM. In mathematical terms:

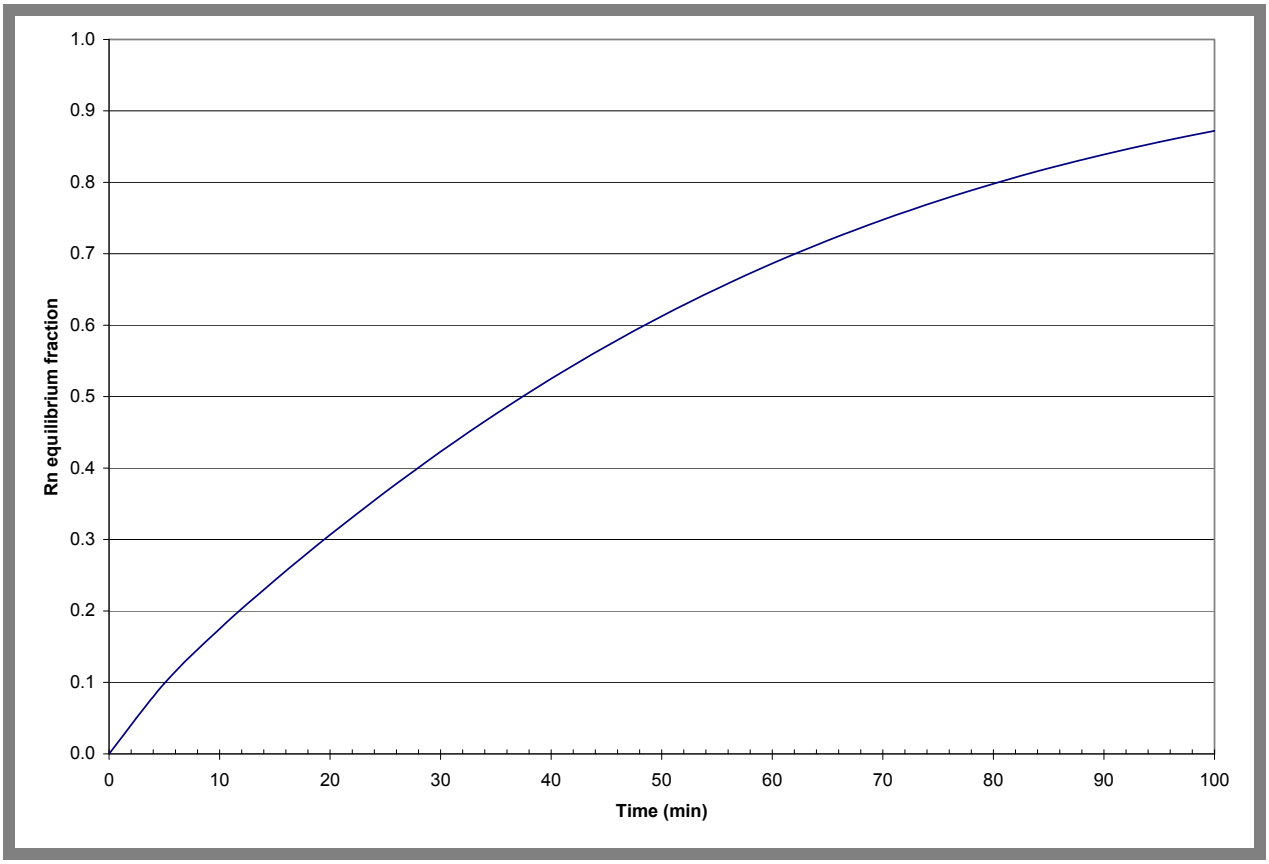
(5.2)

$$\text{Radon exposure (WLM)} = (\text{Rn conc (Bq/m}^3) / 3{,}700 \times F) \times \text{exposure time (h)} / 170$$

where:

WLM = working-level month
Rn conc = radon concentration
F = radon equilibrium fraction

Figure 5.1 Radon Equilibrium Fraction Outdoors



Source: Calculations following procedures in U.S. EPA, 1986.

The estimation of dose from exposure to radon is complicated. The ICRP decided to derive its risk estimates for radon from the results of the many human epidemiological studies of exposure to radon. Based on epidemiological considerations, the ICRP (1993) has estimated that 1 WLM exposure is equivalent on a risk basis to a 4 mSv dose for members of the public (age-independent). Therefore, the dose from radon indoors can be calculated using the equation 5.3 below.

In terms of unit radon concentration in a typical residence (F = 0.4), the DCF using the equation 5.4 below.

Table 5.12 provides a sample calculation of the dose from 1 Bq/m^3 of indoor radon.

Table 5.12 Calculation of Dose from Indoor Radon

Radon Concentration	Exposure	Dose
(Bq/m^3)	Time (h)	(mSv)
1	1	2.5×10^{-6}
1	8,766	0.022
40	7,000	0.70

The last line in Table 5.12 illustrates that someone spending 80% of the year (7,000 h) in a house with a typical radon concentration of 40 Bq/m^3 would be exposed at a dose rate of 0.70 mSv/year.

Another isotope of radon, Rn-220 or thoron, is produced from Ra-224 in the radioactive decay series of natural thorium (Th-232). Similar to Rn-222, thoron also decays to solid decay products with the potential to irradiate the lungs. However, thoron has a relatively short half-life (55 s), and as noted by the ICRP, "The problems posed by radon-220 (thoron) are much less widespread, and generally more tractable, than those posed by radon-222. For protection against thoron, it is generally sufficient to control the intake of the (thoron) decay product, lead-212, which has a half-life of 10.6 hours" (ICRP, 1993).

Therefore, the risk assessor does not normally need to consider thoron in the risk assessment because exposure to thoron and its decay products are expected to be minimal at FCSAP sites.

5.8 Total Dose

The total dose to an individual is referred to as the effective dose and can be calculated using the equation 5.5 below.

In general, it is appropriate to compare the combined dose from external and internal radiation to a dose limit or a reference dose and to compare radon to its own criterion. Hence, in most situations relevant to FCSAP sites, the relevant effective dose would be calculated using the equation 5.6 below.

(5.3)

$$\text{Indoor radon dose (mSv)} = \text{Rn conc (Bq/m}^3\text{)}/3{,}700 \times F) \times \text{exposure time (h)}/170 \times 4 \text{ mSv}$$

(5.4)

$$\text{Indoor radon dose (mSv/h)} = 2.5 \times 10^{-6} \times \text{Rn conc (Bq/m}^3\text{)}$$

where:
Rn con = radon concentration
F = radon equilibrium fraction

(5.5)

$$\text{Effective dose (ED)} = ED_{external} + ED_{internal} + ED_{radon}$$

(5.6)

$$\text{Effective dose (ED)} = ED_{external} + ED_{internal}$$

6.0 DOSE CHARACTERIZATION

6.1 Dose Benchmark

For present purposes, as discussed in Section 2.4, Health Canada proposes a provisional "essentially negligible" dose of 0.01 mSv/year above background for federal sites contaminated with radioactive substances (exclusive of the dose potentially attributable to radon). This level is consistent with the ICRP (2004) recommendation of an annual dose rate of 0.01 mSv as the "minimum constraint" that should be considered for application in any situation.

6.2 Variation in Natural Background Dose

As discussed in Section 2.0, the dose from natural background radiation is quite variable. Given that individuals do not as a rule change their habits or living locations as a result of background radiation, the **variation** in natural background dose might possibly be used as a benchmark against which to evaluate decommissioning or remedial clean-up criteria at contaminated sites. The magnitude of these variations for Canada and the United States are briefly presented below.

The variation in annual doses from natural sources of radiation in Canada has been evaluated by Grasty and LaMarre (2004). They used the annual doses to residents of major Canadian cities to demonstrate the variability in background doses to large groups of the Canadian population. The results of their analyses are shown in part in Table 6.1.

Table 6.1 Average Background Radiation Doses for Canada,Toronto, Winnipeg, and Worldwide

Component	Canada (mSv)	Toronto (mSv)	Winnipeg (mSv)	Worldwide (mSv)
Inhalation (radon)	0.926	(0.757)	(3.225)	1.256
Internal (other than radon)	0.306	0.306	0.306	0.300
Terrestrial gamma	0.219	0.178	0.176	0.48
Cosmic	0.318	0.313	0.315	0.380
Total	**1.769**	**1.554**	**4.022**	**2.416**

Sources: Canada, Toronto, and Winnipeg: Grasty and LaMarre, 2004.
Worldwide: UNSCEAR, 2000.

Table 6.1 represents the variations in **average** background radiation doses in Canada and in two cities. Variations across Canada are wider. For example, based largely on data from a cross-Canada survey of indoor radon levels by Health Canada, Grasty and LaMarre (2004) report the average annual inhalation doses for 16 Canadian cities; they ranged from 0.3 mSv in Vancouver to 3.2 mSv in Winnipeg. Average annual outdoor radiation doses for four Canadian cities, also reported by Grasty and LaMarre (2004), ranged from around about 0.176 mSv in Winnipeg to about 0.240 mSv in Ottawa, but the variability at different locations **within** a particular city is about 10-fold.

The typical background radiation dose in Canada (including the dose arising from exposure to radon) is approximately 2 mSv/year.

The U.S. NRC has examined the concept of background radiation as a criterion for decommissioning (U.S. NRC, 1994a). It evaluated the variations in various components of background radiation, namely inhaled radioactivity (radon), internal radioactivity, gamma radiation, and cosmic radiation. Typical ranges for these major sources of background radiation are provided in Table 6.2.

Table 6.2 Variability of Major Components of Background Radiation in the United States

Component	Typical Minimum (mSv)	Mean (mSv)	Typical Maximum (mSv)
Radon	0.2	2.0	8
Internal (other than radon)	0.3	0.4	1.0
Terrestrial gamma	0.1	0.3	0.8
Cosmic	0.2	0.3	0.6
Overall	**1**	**3**	**10**

Source: U.S. NRC, 1994a.

Overall, according to the U.S. NRC, the range of 1 to 10 mSv (a span of a factor of 10) is typical of the variation in background doses for most people in the United States.

Both the American and Canadian data demonstrate the large variations in natural background doses. UNSCEAR 2000 reports a similar worldwide range in natural background dose.

6.3 De Minimis Dose

De minimis radiation dose is 0.01 mSv/year.

The term "de minimis" originates with the Latin phrase de "minimis non curat lex," which roughly translated means "the law does not concern itself with trifles." Health Canada more commonly uses the analogous term of "essentially negligible" risk. In terms of radiation risk assessment, a de minimis dose is a level of exposure to ionizing radiation considered to pose a negligible or trivial risk, and where the expenditure of additional resources to further mitigate that risk is not justified (CNSC, 2000).

Many jurisdictions and agencies have examined the de minimis concept relative to radiation protection issues. In Canada, the CNSC, and its predecessor the AECB, have looked at the concept in the planning of decommissioning activities (CNSC, 2000), in the application of the ALARA principle to radiation protection (CNSC, 2004), and in the exemption of certain radioactive materials from licensing (AECB, 1989, as cited in CNSC, 2000). Specifically for radiologically contaminated sites, the CNSC Regulatory Document R-104 (AECB, 1987) specifies that at sites where annual doses are less than 0.05 mSv (50 µSv) the dose is acceptable and it is not necessary to remediate the site further to reduce the dose. These sites are not considered likely to cause significant adverse effects on human health. The CNSC further suggests that an annual dose rate of 0.05 mSv (50 µSv) could be considered as de minimis on a case-by-case basis (CNSC, 2000; 2004). Basically, as given in the definition of the term de minimis, the concept is to not waste efforts, either societal or individual, on trivial risks.

The former ACRP (of the AECB and CNSC) also examined this issue and suggested that a risk in the order of 1 in 1,000,000 per year (1×10^{-6}/year) is generally of little or no concern, and that this corresponded to an annual dose rate of about 10 µSv (ACRP/CNS, 1990).

The ICRP, in its recent draft update to its radiation risk coefficients and recommendations, recommends that an annual dose rate of 0.01 mSv (10 µSv) is the "minimum constraint" that should be considered for application in any situation (ICRP, 2004).

A Health Canada report discusses "essentially negligible cancer risk for contaminated site assessment," citing a variety of sources, and notes:

> Although a 1 in 1 million (10^{-6}) cancer risk is the most frequently used risk level for the management of risks posed by environmental (including soil) contamination, many agencies and provinces, including the U.S. EPA, identify a range of increased cancer incidence risks; generally, from 1 in 10,000 (or 1×10^{-4}) to 1 in 1,000,000 (or 1×10^{-6}) is considered an acceptable risk range, depending on the situation and circumstances of exposure (citing Graham, 1993; Kelly, 1991; Lohner, 1997; Travis, 1987; U.S. EPA, 1991) (HC, 2010, p. 59).

and

> In contrast, many industrial standards for workplace environments (e.g. ACGIH, 2002) offer a protection to only the 1×10^{-3} level or higher risk (e.g. a risk of 1×10^{-2}, or 1 in 100, is a 1% chance). This higher cancer risk is "accepted" in workplace environments because it is often

technologically or financially infeasible to reduce exposures to even lower levels, and the nature of exposure is generally deemed to be informed and "voluntary" at the workplace. The U.S. Supreme Court has upheld the industry basis for such standards (citing Graham, 1993)" (HC, 2010, p. 59). Health Canada goes on to note that "CCME (2006) acknowledges that the designation of negligible cancer risk is an issue of policy rather than science, allowing different agencies to establish such a policy consistent with their respective environmental regulatory agendas. To that end, Health Canada, when publishing human health soil quality guidelines in support of the CCME process, applied the concentration of carcinogenic substances in soil associated with risks ranging from 1 to 10,000 (10^{-4}) to 1 in 10,000,000 (10^{-7}) (e.g. citing HC, 1995) (HC, 2010, p. 59).

Health Canada also notes: Although published Health Canada advice on this issue has been restricted to exposures via drinking water, the 10^{-5} risk level has been widely accepted by federal agencies and others involved with contaminated site risk assessment and that the Atlantic Provinces (NB, NS, NFLD/LAB, PEI) have implemented a common approach to contaminated site risk assessment which adopts an acceptable or essentially negligible cancer risk level of 10^{-5} has been adopted (HC, 2010, p. 60).

Health Canada proposes a provisional "essentially negligible" annual dose level of 0.01 mSv above background.

6.4 Radon/Thoron Dose

Indoor radon levels are naturally highly variable from home to home, and levels in individual homes are difficult to reliably predict from the concentrations of Ra-226 in the soil.

Current Radon Guidelines

• Health Canada	200 Bq/m^3
• NORM Guidelines	150 Bq/m^3
• U.S. EPA	148 Bq/m^3
• CNSC (above background)	60 Bq/m^3

The U.S. EPA and Health Canada have similar views with respect to separating radon from other radionuclides. The current Health Canada guideline value is 200 Bq/m^3. The current U.S. EPA gathering value is 4 pCi/L (148 Bq/m^3). Health Canada is currently reviewing Canada's position on this.

The Health Canada NORM Guidelines suggest an *"*unrestricted classification*"* for incidentally exposed workers and members of the public, applied to situations where the average radon concentration (indoors) is below 150 Bq/m^3, consistent with the current U.S. EPA recommendations.

The U.S. NRC in developing its decommissioning guidance commented:

> Because of these variations and the limitation of measurement techniques, the Commission believes that it is not practical for licensees to distinguish between radon from licensed activities at a dose comparable to a 0.25 mSv/y (25 mrem/y) dose criterion and radon which occurs naturally. Therefore, in implementing the final rule, licensees will not be expected to demonstrate that radon from licensed activities is indistinguishable from background on a site-specific basis. Instead this may be considered to have been demonstrated on a generic basis when radium, the principal precursor to radon, meets the requirements for unrestricted release, without including doses from the radon pathway (U.S. NRC, 1997).

The ICRP (1993) suggests that remediation be considered for indoor radon levels of between 200 and 600 Bq/m^3. The U.S. EPA suggests a nominal relationship of 1.25 pCi/L (46.3 Bq/m^3) radon indoors per 1 pCi/g (0.037 Bq/g) radium in the soil. On this basis, a soil Ra-226 level of about 0.17 Bq/g would result in an indoor radon level of about 200 Bq/m^3.

Finally, the CNSC has established a limit of 60 Bq/m^3 of radon above background for sites and activities licensed by the CNSC (Canada Gazette, 2000).

6.5 Gamma Dose Rates

For an Annual Gamma Dose of 0.3 mSv

Duration of Exposure	Gamma Exposure Rate
300 h/year	1 µSv/h
200 h/year	1.5 µSv/h
100 year	3 µSv/h

Uranium tailings contain radionuclide decay and release radiation; when a person walks across a site with tailings, that person is subject to a dose of gamma radiation.[8]

For external gamma radiation, guidance for the radiation dose rate can be established by considering a dose limit and the assumptions about the annual duration of exposure. For example, if one assumes that the annual dose contribution from external gamma will be limited to 0.3 mSv, then the gamma radiation exposure rate limits in the example box can be inferred. It should be noted that 200 h/year is considered to be time spent during casual access to a site. Casual access to a site is defined as people walking across the property during the course of hunting, hiking, skiing, or other recreational activity. Casual access does not include living on a site, working routinely at a site, or gardening at a site.

6.6 Dose Benchmarks for Use at Contaminated Sites Under the Federal Contaminated Sites Action Plan Program

Ideally, federal sites contaminated with radioactive substances (radionuclides) would be remediated to an "essentially negligible" risk level. However, as discussed in this report, there is no consensus as to what such a level should be. Moreover, it is not possible to have "zero" exposure to ionizing radiation and radioactivity. Radionuclides and radioactivity are naturally occurring, are ubiquitous in the natural environment, and vary from place to place.

6.6.1 Background and reference dose

Natural background radiation levels in Canada result in a nominal annual dose of about 2 mSv (highly variable).

Assuming no threshold for radiation risk, this dose rate of natural background radiation could result in a potential lifetime risk of fatal cancer in the order of 2 mSv/year × 70 years × 5 × 10^{-5} ≈ 7.5 × 10^{-3}, or about 1/100.

Health Canada proposes a provisional "essentially negligible" annual dose level of 0.01 mSv above background.

In this context Health Canada (HC, 2003, appendix 2) proposes a provisional "essentially negligible" annual dose level of 0.01 mSv above background for application at federal sites contaminated with radioactivity. This dose rate is approximately equivalent to a lifetime risk of fatal cancer of 5 × 10^{-7} per year of exposure, and a lifetime risk for combined fatal cancers, non-fatal cancers, and serious hereditary effects of 7.3 × 10^{-7} per year of exposure. The corresponding lifetime risk of combined effects would be 5.1 × 10^{-5}.

At federal contaminated sites that have radionuclide contamination, an upper limit of 0.3 mSv above background should not be exceeded for unrestricted land use, per the NORM guidelines. Socio-economic considerations and technical feasibility are important considerations at all contaminated sites. For radionuclides, Health Canada stipulates that the ALARA principle should always be enforced to ensure that human health risks are managed to a minimal level. Levels below 0.01 mSv/year are considered to be negligible and do not require further assessment.

6.6.2 Risk management

The risk assessment should provide information to the various stakeholders who will be considering the results of the risk assessment and other information in making decisions about how to manage the long-term risks from a federal contaminated site.

Section 7.0 discusses the risk levels associated with these dose limits.

6.7 Concentration-Based Remedial Criteria

As discussed previously, the results of an annual dose-based assessment should be compared with a dose limit to determine whether or not additional assessment is needed (see Section 2.4 and Section 6.1). However, it is also possible to "invert" the dose calculation procedure to develop estimates of soil radionuclide levels that could be used to provide a **conservative screen** simply by comparing measured soil concentrations with screening-level soil concentrations. In general terms, the soil screening level could be developed conservatively to reflect the reasonable maximum dose to the critical receptor. If the concentration of a radionuclide in the soil exceeded the corresponding screening value, further

[8] Covers would reduce the gamma exposure rates from tailings with greater reductions for thicker covers; however, any cover material would necessarily contain radionuclides (i.e. uranium and thorium series radionuclides), the gamma exposure rates could never be reduced below background levels.

evaluation is indicated. If on the other hand, the soil concentrations (taken to be representative of the site) were below the screening value, then no further action would be needed.

In addition, if remedial action is indicated as the result of a complex site-specific dose or risk assessment, the site-specific dose equations can be inverted to solve for on-site contaminant concentrations that should reduce exposures to the target acceptable dose level. Various guidances are available from national and international agencies, as described in the following sections. The risk assessor can use these to develop concentration-based remedial criteria.

6.7.1 U.S. Environmental Protection Agency preliminary risk-based guidelines

The U.S. EPA has developed a methodology to develop preliminary risk-based guidelines (PRGs) for application at Superfund sites in the United States (http://epa-prgs.ornl.gov/radionuclides/). The idea is that if the PRG concentration for one or more of the environmental media at the site (including, for example, residential soil, agricultural soil, and fish ingestion among others) are exceeded, then further assessment would be carried out. These PRGs have been derived for a target risk levels of 1×10^{-4} and 1×10^{-6}. PRGs for numerous radionuclides can be obtained from the preceding website.

6.7.2 Health Canada naturally occurring radioactive materials guidelines

The Health Canada NORM guidelines (HC, 2000) provide a table summarizing soil concentrations for unrestricted release of diffuse NORM contaminated areas (i.e. large contaminated areas). The calculated release limits are based on an analysis estimated to result in an annual dose of 0.3 mSv, with a conservative exposure scenario including external gamma radiation, ingestion of soil and vegetation, and inhalation of resuspended contaminated soil. The NORM screening soil concentrations for the U-238 and Th-232 series, both in equilibrium, are reported as 0.3 Bq/g for both decay series. The risk assessor can refer to the summary table (HC, 2000) for other radionuclides.

6.7.3 U.S. Nuclear Regulatory Commission soil-screening limits

The U.S. NRC has published consolidated guidance for decommissioning of radioactive contaminated sites, including screening values of common radionuclides (U.S. NRC, 1994b). The U.S. NRC surface soil-screening values represent surficial soil concentrations of individual radionuclides such that the maximum dose expected from a radionuclide would be 0.25 mSv. Various land use scenarios ranging from agricultural to suburban to commercial/industrial were examined. Because of the very conservative nature of the methodology used to estimate potential exposures and doses, the derived screening limits are such that if the surface soil concentration is below the suggested limits, generally no further action will be required. If the concentration is above the suggested limit, the report recommends that a site-specific dose assessment be conducted before any action is taken.

6.7.4 International Atomic Energy Agency clean-up criteria

The IAEA produced a report for comment on the application of radiation protection principles to the clean-up of contaminated sites (IAEA, 1997). The report examined the conceptual radiation protection framework and the basis for clean-up criteria. It then proposed numeric clean-up criteria in the form of dose limits, and presented the likelihood that clean-up would be needed at these various dose limits. It should be noted that hard and fast limits were not specified, and the concentration limits that would result in specific doses also were not derived.

For purposes of the developing clean-up criteria applicable to range situations, the IAEA considered “bands” of exposure conditions, each covering approximately an order of magnitude in dose or risk. These were numbered from 1 (very low doses less than 10 μSv/year, i.e. 0.01 mSv/y) to 6 (very high doses above 100 mSv/year). The IAEA-proposed clean-up levels are shown in Table 6.3. With constraint and without constraint refer to practices and interventions, respectively.

Table 6.3 Proposed International Atomic Energy Clean-Up Levels

Band No.	Range of Annual Doses (to Average Member of Critical Group)	Is Clean-Up Needed?	
		With Constraint	Without Constraint
6	> 100 mSv/y	always	always
5	10–100 mSv/y	always	almost always
4	1–10 mSv/y	almost always	usually
3	0.1–1 mSv/y	usually	sometimes
2	10–100 µSv/y	sometimes	rarely
1	< 10 µSv/y	almost never	almost never

Source: IAEA, 1997.

The IAEA clean-up levels relate to the annual individual dose to an average member of the critical group in **addition** to the regional background level. This approach is consistent with the Health Canada recommended "essentially negligible" dose where levels less than 0.01 mSv/year do not require clean-up, and at unregulated sites, more than 0.3 mSv/year will usually require some action. The ALARA principle is recommended where levels are between 0.01 mSv/year and 0.3 mSv/year at unregulated sites.

The IAEA (1997) notes that indoor radon levels are very site specific and are often only weakly correlated with the external Ra-226 concentration and, not withstanding the fact that decay includes the dose from radon, that it is "… extremely difficult to distinguish radon attributable to the contamination from background radon levels." The IAEA then states that it would be more sensible, and even necessary also, to **exclude** doses from indoor radon from comparisons with these criteria. This is the approach taken by the U.S. EPA and the U.S. NRC, for example.

7.0 RISK CHARACTERIZATION AND HARMONIZATION

7.1 Radiological Risk Characterization

Annual Risk
Dose × Risk Conversion Factors

Lifetime Risk
Dose × 70 years × Risk Conversion Factors

Risk characterization is not commonly carried out for typical radiological risk assessments. However, for contaminated sites under the FCSAP program, this additional step is recommended. The risk assessor should multiply the intake of radioactivity via inhalation or ingestion by DCFs (this permits the addition of doses from different pathways and to different organs), and add the dose from external gamma radiation to estimate the total effective dose as discussed in Section 2.1 and Section 6.0. The risk is then estimated by risk conversion factors that convert the dose to risk of cancer or hereditary effect.

Nominal ICRP Probability Coefficient for Dose to Annual Risk of Cancer for General Public

- Fatal cancer 5×10^{-5} per mSv;
- Non-fatal cancer 1×10^{5} per mSv;

  Serious hereditary effect 1.3 H 10^{-5} per mSv
- Total 7.3×10^{-5} per mSv

Source: ICRP, 1991.

The ICRP uses the term "detriment" to represent the combination of the probability of occurrence of a harmful health effect and a judgment of the severity of that effect. The many aspects of detriment make it undesirable to select a single quantity to represent the detriment, and the ICRP (1991) has adopted a multi-dimensional concept. The principal components of detriment include the following stochastic quantities: the probability of attributable fatal cancer, the weighted probability of attributable non-fatal cancer, the weighted probability of severe hereditary effects, and, the length of life lost if harm occurs. The values of this aggregated detriment at low doses for both the exposed working population and the exposed general population are given in the ICRP recommendations (ICRP, 1991).

The risk from radiation exposure is determined by use of the risk coefficients given by ICRP and the calculated effective dose equivalent for an individual. For regulatory purposes, it is generally assumed that the risk increases with increasing dose in a linear fashion and that zero dose corresponds to zero risk (i.e. linear non-threshold dose response relationship).

Lifetime risk of all cancers and serious hereditary effects for 1 mSv/year:

$$\left(\frac{1\ \text{mSv}}{\text{year}}\right) \times (70\ \text{years}) \times \left(7.3 \times 10^{-5}/\text{mSv}\right) = 5.1 \times 10^{-3}$$

7.2 Context

As with any naturally occurring substance, it is possible that substantial exposures could occur in the environment. The risk assessment of substances that can cause cancer (carcinogens) at contaminated sites considers the incremental risk of getting cancer from exposure to the substances found at that site compared with the background risk of cancer in the population. Common environmental risk assessment practice in Canada considers that an incremental cancer risk of 1 in 100,000 (1×10^{-5}) is essentially negligible compared with the typical background risk of getting cancer in the general population.

To put the risks into perspective on "essentially negligible" or "de minimus" dose of 0.01 mSv/year, this dose is associated with a lifetime risk of all cancer and serious hereditary effects at the rate of 3.5×10^{-5} that is incremental above background.

Incremental Risk

Lifetime risk of all cancer and serious hereditary effects for de minimus dose of 0.01 mSv/year:

$$\left(\frac{0.01\ \text{mSv}}{\text{year}}\right) \times (70\ \text{years}) \times \left(7.3 \times 10^{-5}/\text{mSv}\right) = 5.1 \times 10^{-5}$$

The potential risk of getting cancer from lifetime exposure to natural background radiation can be calculated in a similar fashion. Natural background radiation levels in Canada result in a nominal annual dose of about 2 mSv/y. The lifetime risk of fatal and non-fatal cancers and serious hereditary effects arising from exposure to 2 mSv/year are about 1.0×10^{-2} (about 1/100).

Background risk

Lifetime risk of all cancer and serious hereditary effects from 2 mSv/year (nominal background):

$$\left(\frac{2\ \text{mSv}}{\text{year}}\right) \times (70\ \text{years}) \times \left(7.3 \times 10^{-5}/\text{mSv}\right) = 1.0 \times 10^{-2}$$

Between 25% and 30% of Canadians die from cancer; thus, given the above assumptions, **natural** background radiation theoretically accounts for about 4% of background (fatal) cancers (i.e. 1/100 cancers attributable to natural background radiation compared with a rate of fatal cancer in the population, from all causes, of about 25/100).

7.3 Harmonization of Chemical and Radiological Risks

Within the regulatory framework for risk assessment, there are several differing regulatory policies for limiting routine exposures of the public to radionuclides and other chemical carcinogens (Overy and Richardson, 1995). In particular, there are inconsistencies in the level of acceptable health risks associated with radionuclides as developed by different regulatory agencies (e.g. Kocher and Hoffman, 1991). Historically, radiation exposure in North America was considered not to be regulated as stringently as chemical exposure (Travis et al., 1989). However, an analysis of risk management practices for radiation and chemicals has determined that risk management strategies for both hazards are well developed and similar in principle, and that both risk management strategies provide a high degree of health and environmental protection (HC/AECB, 1998).

The ICRP (1991) recommends limiting annual above-background radiation exposure to the general public to 1 mSv. This corresponds to a lifetime (70 years) fatal cancer risk of about 4.3×10^{-3}. For risk assessment of chemicals at federal contaminated sites, Health Canada identifies a target incremental cancer risk level of 1×10^{-5} for chemicals that affect the same target organ for both fatal and non-fatal cancer (HC, 2003). The U.S. EPA, in its guidance on radiation risk assessment (U.S. EPA, 1989, chapter 10) and supplemental guidance on radiation risk assessment (U.S. EPA, 1989), emphasized that risk characterization and remedial goals for radioactive substances at National Priority List (Superfund) *Comprehensive Environmental Response, Compensation, and Liability Act (CERLA)* sites must target the 10^{-4} to 10^{-6} risk range for cumulative cancer risk. In particular, the U.S. EPA stipulates:

> Guidance issued by other organizations (eg., NRC, DOE, ICRP, NCRP) may provide technical assistance, however the reader should exercise caution since some of these documents utilize a framework for risk management (eg., allowable dose limits of 25, 100 or 500 mrem/yr) that the U.S. EPA has determined is not suitable for use at CERCLA sites (U.S EPA, 1999, p. 2).

Both the U.S. EPA and the IAEA propose clean-up criteria that are consistent with the target incremental cancer risk level for chemicals at federal contaminated sites in Canada. It is recognized that technical feasibility and costs must also be factored into remediation of contaminated sites. Federal contaminated sites in Canada are more closely aligned with Superfund (CERCLA) sites in the United States with respect to the types of sites and nature of exposed populations (predominantly the general public). The Canadian FCSAP program, although addressing federal sites only, is more similar in its goals to the American Superfund (CERCLA) program than to other programs that regulate radiological exposures.

Because radionuclides and chemical carcinogens often coexist, it is important to identify and analyze these differences, and to establish a practical approach to defining an acceptable level of exposure. Furthermore, in developing risk criteria, it is important to understand the inherent uncertainties underlying the risk assessment of chemical and radioactive carcinogens (e.g. Chambers et al., 1987; Hoffman et al., 1993; Goldammer, 1995; Hrudey, 1996).

An assessment of differences in uncertainties and confidence levels in the risk assessment of chemical and radioactive carcinogens include differences in background levels and natural variability; different degrees of uncertainty in risk characterization; different endpoints and definitions of risk factors (e.g. use of upper 95th percentile in some cases and expected values in others); and the use of reasonable maximum exposure versus average, typical, or population-wide exposures. At some sites, it may not be technically possible to determine with accuracy the background concentration to a level that is consistent with the proposed "essentially negligible" value. Exposure to levels above 0.01 mSv may be considered, based on technical feasibility of monitoring and/or remediation of the site. The ALARA principle should be considered in radiological risk assessment for exposure to levels between 0.01 and 0.3 mSv at unregulated federal contaminated sites. Levels above 0.3 mSv at unregulated federal contaminated sites must be discussed with stakeholders.

The EPA Science Advisory Board has commented:

> While cancer risk estimates for radiation entail substantial uncertainties, especially at low doses and dose rates, they are seen as being sufficient to justify making a best estimate of risk within a statistical uncertainty factor of about 2 for all cancers combined for whole-body external radiation

> if the dose is known accurately (NCRP 1989). These best estimates of risk are used directly without further safety factors of any kind. Because best estimates are used and the degree of uncertainty is only moderate, risk assessment results for radiation can be compared with risk criteria for control decisions with some confidence (U.S. EPA/EPA SAB, 1992).

The EPA Science Advisory Board also suggested that this is quite different from chemical risk assessment where:

> To deal with the uncertainty, EPA in particular has adopted the use of the upper confidence limit on the slope of the linearized multistage model to project risks at low doses and has used a conservative procedure - the surface area scaling rule - to project from animal bioassays to assumed human responses. Both of these procedures are widely believed to produce risk estimates that are more likely to overestimate than underestimate human risk. Thus risk estimates for chemicals are biased high (even though such may not be the case with every chemical). This conservative method of dealing with uncertainty ensures that in the vase majority of cases, the actual risk level achieved will be lower than the risk criterion used in a control decision (U.S. EPA/EPA SAB, 1992).

As discussed previously, a practical difficulty is associated with harmonization. The problem is whether to and how to combine risks from a given exposure to a variety of chemical and radiological hazards into a single number to support a decision regarding clean-up. Risk factors for chemicals are uncertain and tend to be conservative (biased high). If used as they are presented for a single contaminant, there can be quite a high degree of conservatism. When several contaminants are concurrently evaluated, the conservatism has potential to become extreme (assuming independence between the risk factors) leading to highly skewed results that are inappropriate for cost-benefit analysis (Pollock et al., 1995).

Philosophically, it can be argued that it is appropriate to worry more about the risks that are not well known. Conversely, it is possible to argue that risk factors for the most toxic or carcinogenic substances are generally known better than for the less toxic substances. Because the more toxic substances are studied more intensively, more often than not quantitative data (toxicological or epidemiological) are more available for these substances. However, the overall uncertainty in a risk estimate depends not only on the risk factors, but also on uncertainties in other parameters such as receptor characteristics, biological transfer factors, etc.

Currently, several conflicting regulatory policies exist for limiting routine exposures of the public to radionuclides and chemical carcinogens. Some of these differences are due to historical misconceptions (Rosenthal et al., 1992). Others are related to scientific/technical differences – such as different receptor characteristics, different environmental fate and transport models, different biological transfer factors, etc. – all of which are equally defensible on scientific grounds. The divergence of radiological and chemical risk assessments has evolved primarily owing to the separate and divergent maturity of the two disciplines of health physics and chemical risk assessment. Therefore, it is a challenge to harmonize radiological and chemical risk assessment. Notwithstanding these differences, the concept that similar risks should be treated alike is logical (Pollock et al., 1995), and should be pursued. Kocher suggests that the differences in the regulatory approach for radionuclides and cancer-causing chemicals can be reconciled by "adapting consistent and reasonable usage of the terms 'acceptable' and 'unacceptable' and in describing risks and recognizing that application of the objective that exposures should be as low as reasonably achievable (ALARA) is the most important factor in determining acceptable risks for any exposure situation" (Kocher, 1999).

Health Canada is pursuing greater harmonization of chemical and radiological risk assessment. Almost two decades have passed since the U.S. EPA/EPA Science Advisory Board review report (U.S. EPA/EPA SAB, 1992). The reported impossibility of bringing the two areas into conformity in 1992 is now, in fact, possible, at least to a limited degree. To that end, Health Canada has harmonized radiological and chemical risk assessment for Canadian federal contaminated sites in the following ways:

1. The "essentially negligible" risk level to be applied to the characterization of risks from federal sites contaminated with radiation is considered to be 0.01 mSv. Although the lifetime risk from this incremental dose level is 5.0×10^{-5}, exceeding the 1 in 100,000 (1 10^{-5}) essentially negligible lifetime risk applied to chemical carcinogens, it represents a significant step in reducing the large discrepancy between radiological and chemical risk assessment benchmark risks. However, it is noted that when significant technical restraints are associated with meeting this level of exposure, the ALARA principle should be employed, and levels should not exceed 0.3 mSv at unregulated federal sites.

2. Receptor characteristics for radiological and chemical risk assessment have been harmonized based on available Canadian and North American data. The data most representative of the Canadian population are the most valid and defensible basis for the assessment of risks

posed to Canadians, whether for chemicals or for radionuclides.

3. Fate and transport models, and biological transfer factors (for predicting uptake into plants, fish, livestock, and game) to be employed in both chemical and radiological risk assessments for Canadian federal contaminated sites will be consistent and comparable. To this end, Health Canada is conducting various evaluations to determine which fate and transport models will be most appropriate for application within the FCSAP program.

It should be noted that many aspects of the components of radiological risk assessment will be adopted within the Health Canada chemical risk assessment guidance to effect revisions to existing standards of practice in that discipline. Standards of practice for radiological risk assessment will not be the only procedures to be revised.

These steps, although minor, will significantly increase the harmonization between radiological and chemical risk assessment in Canada. These steps will also significantly improve the ability of risk assessors to communicate the results of radiological risk assessments to affected communities; such communities often do not comprehend why current radiological risk assessment appears to provide less health protection than chemical risk assessment.

8.0 ADDRESSING UNCERTAINTY

Uncertainty is unavoidable in dose and risk assessments, and it is important for the risk assessor to identify potential sources of uncertainty and the likely effect that uncertainty has on the estimates of dose and risk. As a minimum, a dose/risk assessment should provide a qualitative discussion of uncertainties. Qualitative discussions are particularly appropriate for screening-level assessments. For complex site-specific risk assessments and particularly for probabilistic assessments, it will be desirable to quantitatively evaluate the effect of uncertainty and provide selected statistical attributes of dose/risk (e.g. the mean, the standard deviation of the mean, the 95th confidence interval about the mean, various percentiles of estimated dose/risk). Quantitative assessments of uncertainty will include the analysis of how sensitive the results of a dose/risk assessment are to a particular assumption or parameter – i.e. how much influence a parameter exerts on the dose estimates (a sensitivity analysis). Sensitivity analyses can be useful in helping to assess the need for further data collection, particularly if one or more influential parameters have not been measured directly on site, and to focus such efforts.

It is the responsibility of the risk assessor to determine the appropriate probability distribution function to use.

This guidance manual does not provide a step-by-step procedure for probabilistic uncertainty analysis, but this section is intended to provide an overview of key issues. In addition, suggested probability distributions for a number of selected parameters of possible interest in assessing uncertainty for a site of interest are provided in appendices A and B. Richardson (1997) provides suggested probability distribution functions for a variety of receptor characteristics based on data for the Canadian population. It is emphasized that these distributions are provided only as a possible starting point and that it is the responsibility of the risk assessor to decide what distributions are most appropriate and defensible for the probabilistic assessment at any contaminated site.

There is a great deal of discussion concerning the characterization and analysis of uncertainty in the literature, and the reader interested in this topic is referred to the numerous documents and scientific papers that are available. The following documents are considered to provide a good introduction to the subject matter, but should be recognized as representing only a small sample of the available literature. It is the obligation of the risk assessor in developing an approach to uncertainty analysis to be aware of current literature on this subject.

Suggested Background Reading

BIOMOVS II 1993. *Guidelines for Uncertainty Analysis.* Technical Report No. 1. July.

Hoffman, F.O. and J.S. Hammonds. 1992. *An Introductory Guide to Uncertainty Analysis in Environmental Health and Risk Assessment.* Oak Ridge National Laboratory, United States Department of Energy.

International Atomic Energy Agency (IAEA). 1989. *Evaluating the Reliability of Predictions Made Using Environmental Transfer Models.* IAEA Safety Series 100.

United States National Council on Radiation Protection and Measurements (U.S. NCRP). 1996. *A Guide for Uncertainty Analysis in Dose and Risk Assessments Related to Environmental Contamination.* NCRP Commentary No. 14.

United States Environmental Protection Agency (U.S. EPA). 1997. *Guiding Principles for Monte Carlo Analysis.* EPA/630/R-97/001, March.

United States Environmental Protection Agency (U.S. EPA). 2000a. *Options for Development of Parametric Probability Distributions for Exposure Factors.* EPA/600/R-00/058, July.

8.1 Sources of Uncertainty

Several sources of uncertainty are described below:

- **Parameter uncertainty –** includes natural variability in data (such as in concentration measurements), natural inter-individual variability (such as in breathing rates, water and food consumption rates, metabolic rates, etc., measurement errors, sampling errors, and systematic errors
- **Model uncertainty –** includes such considerations as whether the model is valid, if it is a simplification of real-world processes, model misuse, or use of inappropriate surrogate variables
- **Scenario uncertainty –** involves descriptive errors, aggregation errors, errors in professional judgment, and incomplete analysis. (U.S. NCRP, 1996; U.S. EPA, 1997; Bartlett et al., 1996)

In general terms, for dose and risk assessment, an uncertainty assessment can encompass two broad categories:

- uncertainty – considered as a lack of knowledge
- variability – considered as the natural variation in the parameters in question

Uncertainty and variability can both be measured or quantified by probability distributions. It can be argued that all probabilities may be interpreted as subjective probabilities. The selection of probability distribution functions is another aspect of uncertainty (Bartlett et al., 1996). Thus, the development of probability distribution functions can be used without distinction between whether the distribution arises from uncertainty or variability, or whether the basis for the distribution is subjective or empirically (data) driven. Whether uncertainty and variability are dealt with collectively or separately depends on the assessment question(s) being asked. The interpretation of the analysis will depend on how the question(s) was asked and how the "uncertainty" was managed in the analysis (IAEA, 1989; NCRP, 1996).

In broad terms, the sources of uncertainty that occur in the conceptual model and risk assessment include:

- incomplete or incorrect definition of the **conceptual site model** – For example, have all relevant pathways been identified? As noted in Section 3.0, the problem formulation stage of the assessment is key, and the relevance and quality of the subsequent dose/risk assessment is closely linked to the care given to problem formulation.
- formulation of the **mathematical description** of the conceptual site model – These uncertainties can be tested (validated) against available site data when site data are available. Data management, numerical implementation and programming errors are also possible and need to be managed through a quality process.
- uncertainties in data, assumptions, model parameters, and the selection of probability distribution functions to represent data distributions.

8.2 Evaluating Uncertainty

A great many approaches to addressing uncertainty arise in exposure pathways. Dose and risk assessment are discussed in the open literature, and many publications address the ways in which uncertainty associated with key assumptions and model parameters can be propagated through mathematical models to develop (subjective) probability distributions for various model inputs and outputs, such as concentrations of radionuclides in environmental media, doses, and risks. A variety of methods, both analytical and using Monte Carlo simulation (referred to in this manual as probabilistic), are available. The U.S. EPA states "that such probabilistic analysis techniques as Monte Carlo analysis, **given adequate supporting data and credible assumptions**, can be viable statistical tools for analyzing variability and uncertainty in risk assessments" (U.S. EPA ,1997, p.1).

Steps Leading to an Uncertainty Analysis

- problem formulation
- conceptual site model
- deterministic (reasonably conservative) analysis
- qualitative uncertainty analysis
- semi-quantitative sensitivity analysis
- probabilistic analysis, predicting range and distribution of doses/risks for entire receptor population

The text box outlines the steps leading to a probabilistic analysis. As discussed in the next section, decisions may be based on the outcome of the dose/risk assessment; therefore, it is important for such assessments to provide assurance that the estimated dose and risks are scientifically defensible.

In addition, when conservative screening assessments indicate that a reference dose or risk is exceeded, the exceedances may in fact be an artifact of worst case or RME assumptions. A probabilistic analysis can provide predictions of the range and distribution of dose and risk across the entire receptor population, and provides significant insight as to the "reasonableness" of the estimated doses/risk and the "amount of conservativeness" in the assessment.

8.3 Probabilistic Uncertainty Analysis

8.3.1 Monte Carlo analysis

Various techniques are available, as indicated earlier; however, the most common method of conducting a probabilistic assessment is to employ Monte Carlo simulation. In this type of assessment, probability distribution functions that reflect various types of uncertainty are assigned to all input parameters. The following calculation sequence is then implemented:

- Draw random samples from the selected probability distribution.
- Enter the selected values into the mathematical model.
- Save the predicted effect.
- Repeat many times to develop a subjective probability distribution of effect.

The risk assessor must decide whether or not a probabilistic analysis is needed. As general guidance, it is suggested that:

- no further analysis (deterministic or probabilistic) needed if dose predicted to be <0.01 mSv/year ("essentially negligible" risk)
- follow-up analysis possibly involving data collection, sensitivity analysis, refined assumptions for dose <0.15 mSv/year
- follow-up analysis possibly involving additional data collection, refined use of model and probabilistic analysis indicated for dose > 0.15 mSv/year

This model simply involves multiplying the three parameters together. Alternatively, it may be a very complex model, where many different parameters are used (e.g. to describe the solubility of a radionuclide in mine tailings). A Monte Carlo analysis can include a mix of distributions and point estimates for the input parameters to the model. In any event, the calculation sequence would be the same, as shown above.

The "set" of output values from such an analysis can be considered as a sample on an objective random variable, and any desired statistical attribute of that sample, such as mean, variance or full probability function, can be estimated. In essence, the output of such an analysis can be considered as a (subjective) probability distribution function, which describes in quantitative terms what we are entitled to believe about the output variable. However, the confidence in the results is clearly dependent on the appropriateness of the conceptual site model and the confidence in the data, mathematical model, model parameters, and other assumptions underlying the analysis.

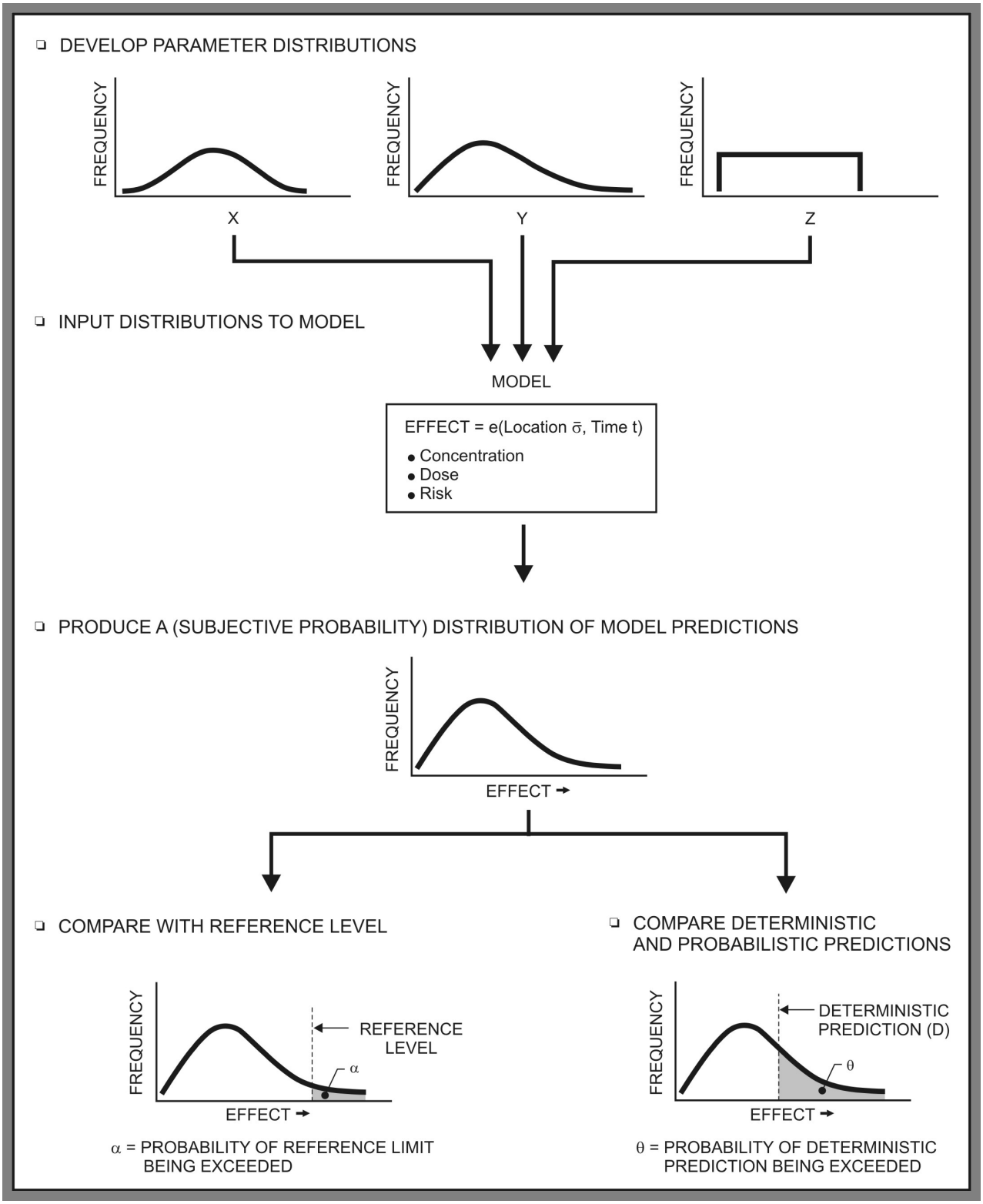
There are two common random sampling processes in Monte Carlo analysis: Simple Random Sampling and Latin Hypercube Sampling. In Simple Random Sampling, a random number is selected from the distribution; the probability of a value being chosen is determined only by its PDF. In Latin Hypercube Sampling, sampling may be viewed as a stratified sampling scheme designed to ensure that the upper or lower ends of the distributions used in the analysis are well represented. Although not considered an endorsement, two commonly available models Crystal Ball® and @Risk are available for use with spreadsheet calculations to perform uncertainty analysis, and both models have been widely used for this purpose.

According to Vose's *Quantitative Risk Analysis: A Guide to Monte Carlo Simulation Modelling*, the cardinal rule of risk analysis modelling is "Every iteration of a risk analysis model must represent a scenario that could physically occur" (Vose, 1996,). Following this rule will lead to a model that is both accurate and realistic.

Figure 8.1 illustrates the concept of probabilistic analysis. An effect model, for example, can be a simple model such as:

intake of radionuclides from fish = [concentration of radionuclide in water "X"] × [transfer factors from water-to-fish "Y"] × [consumption of fish "Z"]

Figure 8.1 Concepts of Probabilistic Analysis



8.3.2 Commonly used probability density functions

In a quantitative uncertainty assessment, probability density functions can be used to define the uncertainty in parameter values. Many distributions can be used to describe the variability in a parameter; however, some of the most common and useful distributions are as follows:

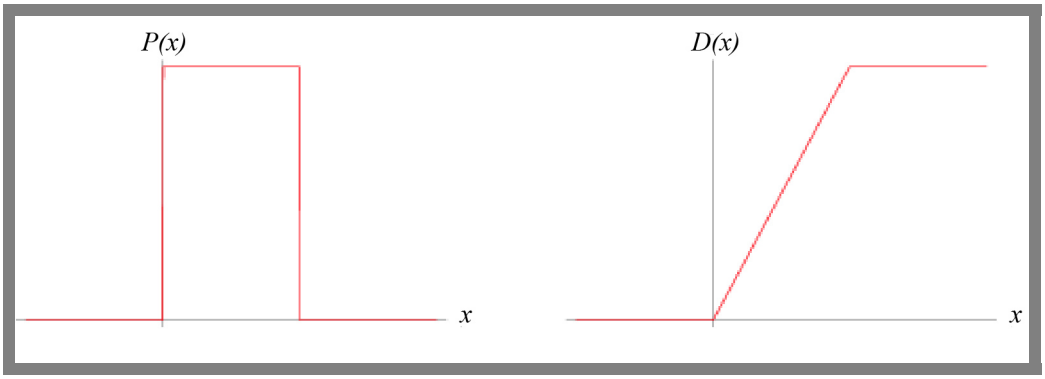
- constant
- uniform
- triangular
- normal
- log-normal
- beta

8.3.2.1 Constant

A constant value is appropriate for well-known parameters with small uncertainty.

8.3.2.2 Uniform

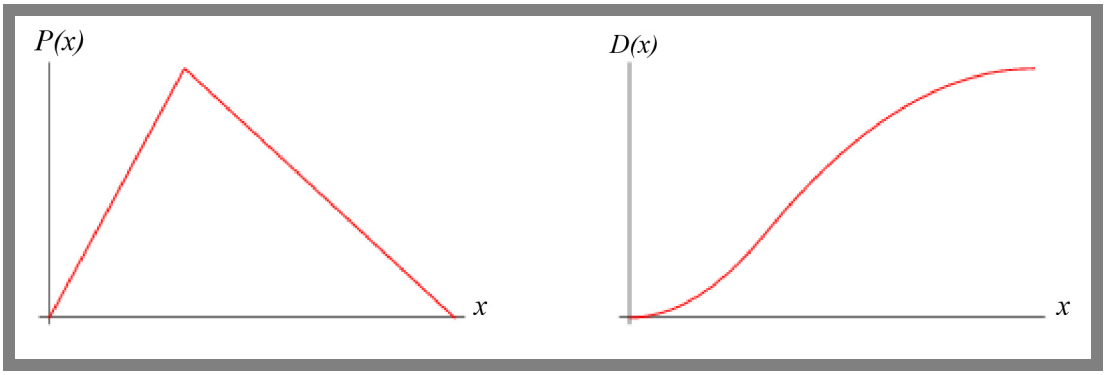
A uniform distribution can be used when there is some knowledge of range of values, but a minimal understanding of how often a value would occur. There is an equal probability that any value between the specified minimum and maximum would be selected.



Source: http://mathworld.wolfram.com/.

8.3.2.3 Triangular

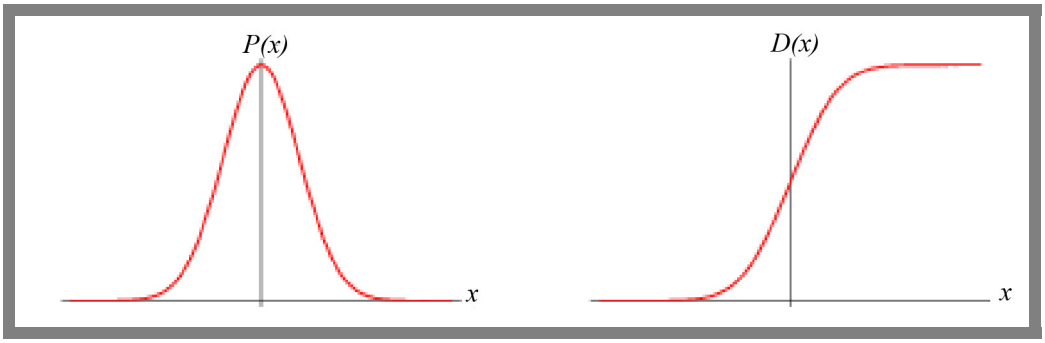
A triangular distribution can be used when there is knowledge of the range of values and some understanding of the shape. A triangular distribution is specified by its minimum, maximum, and mode values. For a symmetric distribution, the mean of a distribution is specified as the mode. However, it does not have to be symmetric, and can be skewed either to the left or right.



Source: http://mathworld.wolfram.com/.

8.3.2.4 Normal

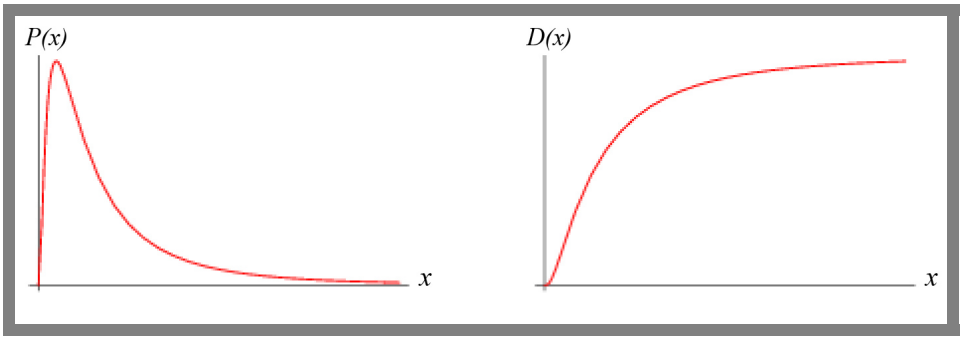
A normal distribution is useful for symmetrical data. For a normal distribution, 99.73% of all samples will fall within three standard deviations of the mean value.



Source: http://mathworld.wolfram.com/.

8.3.2.5 Log-normal

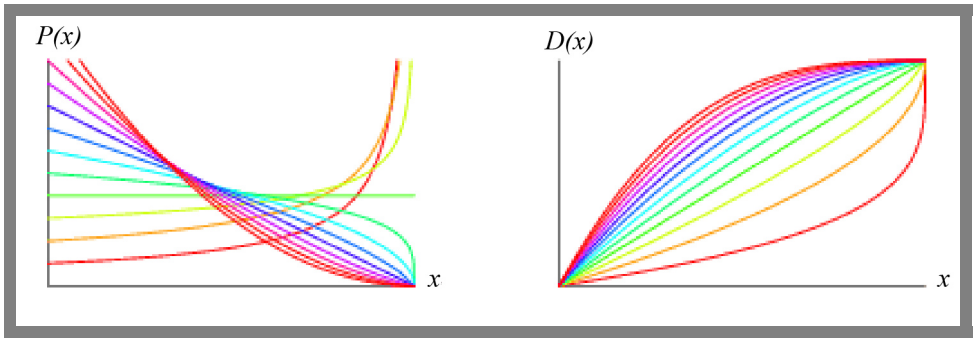
A log-normal distribution is a continuous distribution in which the logarithm of a variable has a normal distribution. This distribution is appropriate for application to a skewed data set when a considerable amount of data is within a small range. The log-normal distribution can be used only for variables that are always positive. This distribution type is widely used in environmental applications.



Source: http://mathworld.wolfram.com/.

8.3.2.6 Beta

A beta distribution is useful as it is a bounded distribution that is able to mimic many distributions by changing the shape parameters. The following plots illustrate some of the flexibility of the beta distribution; α is set to 1 and β ranges from 0.25 to 3.00.



Source: http://mathworld.wolfram.com/.

8.3.3 Assigning a probability density function

The risk assessor should consider several methods in defining a probability distribution function. First, the choice of input distribution should always be based on considerations of all information (both qualitative and quantitative) available for a parameter (U.S. EPA, 1997). The probability distribution function can then be assigned by:

1. Fitting the data to a distribution. It should be noted that this is generally a subjective process.
2. Given a data set it is possible to use a statistical program (e.g. SAS) to estimate parameters that describe probability distribution functions (e.g. maximum likelihood method, goodness of fit).
3. Adoption of standard probability distribution functions (many have been developed for exposure parameters (e.g. Richardson, 1997; U.S. EPA, 2000) (see Appendix B). This adoption still requires expert guidance on the appropriateness of the probability distribution function (e.g. include consumption rates for eaters and non-eaters).
4. Subjective probability distribution functions can also be developed using professional judgment and available data. A rule of thumb is that if a parameter can be expressed as a quotient of other variables, it is often possible to approximate its probability distribution function with a log-normal distribution (e.g. the 1993 BIOMOVS II publication).

Cautions

- It is important to specify if a parameter is correlated to another parameter.
- It is important to document the basis for probability distribution functions in the assessment as well as the supporting information or development procedure.

8.4 Interpretation of Results

Risk assessments are used to provide information to assist those responsible for making decisions about what, if any, remedial actions should be taken at a particular contaminated site. As discussed previously, intentionally conservative dose and risk assessments (i.e. a screening analysis) are performed early in the risk assessment. If the predicted dose or risk is below the corresponding dose or risk objectives, then no further analysis is needed. On the other hand, a dose or risk predicted to exceed the corresponding objectives does not necessarily imply a hazard exists, but rather points to the need for additional analysis or investigation. Probabilistic analyses offer one approach for developing an improved understanding of what the doses (and risks) may be and, importantly, how confident one can be about them.

As discussed by Hoffman et al. (1999, p. 256), "The move toward explicit recognition of uncertainty has been acknowledged by regulatory agencies such as the EPA" which notes "Reliable information may or may not be available for many aspects of a risk assessment. Scientific uncertainty is a fact of life for the risk assessment process…." These authors further note that "Regulations, at least those with which we are familiar, are couched in terms of what is often referred to as a bright line, simply put, regulations are typically formulated so that a parameter (e.g. concentration, dose or risk) is either below a pre-specified number or above it. If the value is below the regulatory limit, then by definition we are in compliance with the regulation; if the value is above the limit, we are in violation" (p. 256).

The mean dose and the upper 95th confidence level in the mean dose (or the upper 95th percentile of the distribution) are common attributes that are used in probabilistic assessments. In a probabilistic assessment, the attributes used to interpret the results should be discussed in the problem definition phase.

Although it is straightforward to compare the results of a deterministic analysis to reference dose or risk levels and conclude that the predicted dose is above or below the reference level, the interpretation of a probabilistic uncertainty analysis (i.e. that is one which explicitly attempts to accommodate uncertainty arising from lack of knowledge and variability) is more complex. Consider, for example, the sketch in the lower part of Figure 8.1 to illustrate the results of an uncertainty analysis. The smooth curves shown in Figure 8.1 might illustrate the probability of an exposed individual experiencing dose D within the limits of the scenario, models, and assumptions of the analysis. The reference dose level is shown. For a probabilistic analysis, the average or expected dose, the median calculated dose, or some other attribute (e.g. the upper 95th percentile) could be compared with the reference dose level. Finally, the chance that the exposed person receives a dose larger than the reference dose is shown as the shaded area α. The area (1-α) then represents the "degree of belief which we are entitled to hold" that the actual dose will not exceed the reference dose.

It is often informative to compare both the mean predicted dose and the upper 95th confidence level of the mean or the upper 95th percentile of the distribution to the reference dose.

Also shown at the bottom part of Figure 8.1 is the location of the deterministic dose compared to the percentile distribution of dose. The relation of the (conservative) bounding deterministic dose to both the mean dose and some percentile (such as the 95th percentile) can be informative.

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10.0 GLOSSARY

Absorbed dose: In chemical exposure assessment, the amount of a substance that penetrates an exposed organism's absorption barriers (e.g. skin, lung tissue, gastrointestinal tract) through physical or biological processes. The term is synonymous with **internal dose**. In radiological assessments, the amount of energy deposited by ionizing radiation per unit mass (see **gray**).

Activity: The rate of disintegration of a radioactive substance, which is a measurement of the number of becquerels of a radioactive species in a sample.

Alpha radiation: The least penetrating, but most strongly ionizing, of the three principal forms of radiation from radioactive materials. Alpha radiation can be stopped by the outer layer of dead skin cells in human skin or by a single sheet of paper. However, alpha radiation can damage live body cells if ingested or inhaled through food, water, air, etc.

Ambient air: Any unconfined portion of the atmosphere: open air, surrounding air.

As low as reasonably achievable (ALARA): A concept in radiation protection according to which radiation exposures are kept as far below the regulatory limits as possible, taking into account the state of technology achievable and the cost of improvement in relation to (1) benefit or risk to the environment and to public health and safety; (2) other societal and socio-economic considerations; and (3) the use of radioactive materials in the public interest in medical diagnosis and therapy, research, the manufacturer of consumer products, and the production of electricity by nuclear power reactors.

Atmospheric dispersion: The aerial release/distribution/dilution/migration of a substance into the atmosphere of matter and gases that can be carried by air currents.

Background radiation: The radiation in the natural environment, including cosmic rays and radiation from naturally occurring radioactive elements. It is also called natural radiation.

Becquerel or Bq: A standard international unit of radioactivity, equal to one radioactive disintegration per second. The obsolete unit curie or Ci, based upon the amount of radioactivity in a gram of radium, equals 3.7×10^{10} Bq.

Best management practice (BMP): Methods that have been determined to be the most effective, practical means of preventing or reducing pollution from non-point sources.

Beta Radiation: Beta particles are electrons ejected at high speeds from the nucleus of an atom undergoing radioactive decay to a proton and an electron. Beta particles are ejected from the nucleus over a continuous energy spectrum. Unshielded beta sources can constitute external hazards if the beta radiation is within a few centimetres of exposed skin surfaces and if the beta energy is greater than 70 keV. Internally, beta particles have a much greater range than alpha particles in tissue. However, because they cause fewer ionizations per unit path length, beta particles deposit much less energy to small volumes of tissue, and consequently inflict less damage than alpha particles.

Bioaccumulation: The net accumulation of a chemical by an organism as a result of uptake from all routes of exposure.

Bioavailability: The degree to which a substance is available to be absorbed and metabolized by an exposed living organism.

Carcinogen: An agent that has the potential to cause cancer.

Chronic effect: An adverse effect on a human or animal in which symptoms recur frequently or develop slowly over a long period of time.

Chronic exposure: Multiple exposures or continuous exposure occurring over an extended period of time or over a significant fraction of an animal or human lifetime (U.S. EPA defines this as 7 years to a lifetime).

Chronic toxicity: The capacity of a substance to cause adverse health effects in humans, animals, fish, and other organisms, as a consequence of chronic exposure.

Compliance monitoring: Monitoring of operations to ensure they comply with government regulatory standards and requirements.

Composite sample: A sample of soil or other environmental media for analysis that is made up by combining two or more individual samples from the same sample site.

Conservative: As used in the term "conservative estimate," it is considered to be an overestimate of the actual concentration/exposure/dose/risk, effect or hazard being assessed; it can also be interpreted as "unlikely to have been underestimated."

Contaminant migration: The movement of contaminants from one location to another via diffusion, dispersion, by currents, flows, gravitational influence, etc.

Contamination: The presence of substances, both radioactive and non-radioactive, at levels (concentration, mass, etc.) above those normally or naturally found (i.e. above background) and that may cause an adverse effect.

Contingency plan: A pre-arranged plan to be implemented in the (unlikely) event of some possible accident, whether foreseen or unforeseen, of serious concern.

Curie: See **becquerel**.

Decay: The disintegration of the nucleus of an unstable radionuclide by the spontaneous emission of energy or particles, resulting eventually in a decrease in the number of radioactive atoms in the sample.

Decay chain: The series of radionuclides that form sequentially as radioactive decay progresses before reaching a stable form (e.g. the U-238 decay chain contains 14 radionuclides starting with U-238 and ending with stable Pb-206).

Decommissioning: The act of removing a regulated facility from operation and operational regulation. This usually entails a certain amount of clean-up (decontamination).

Dose: See **effective dose** (unless otherwise specified).

Effective dose: This term is intended to express radiation doses in a manner such that the long-term biological harm to humans will be approximately the same per unit of effective dose, regardless of the type of radiation involved or of the parts of the body exposed to radiation. To obtain effective dose in Sv, the absorbed radiation dose in Gy is multiplied by the appropriate radiation factor and, in the case of partial body exposure, by the appropriate tissue weighting factors. Both of these factors are taken to be one in the case of whole body exposure to gamma rays. Further details can be found in ACRP-13 (1991) and ICRP Publication 60 (1991).

Exposure: The amount of ionizing radiation that strikes a living or inanimate material. In health physics, exposure is more specifically defined as a measure of ionization in air caused by X-ray or gamma radiation only.

Exposure assessment: Identifying the pathways by which radionuclides and radiation may reach individuals, estimating how much radiation an individual is likely to be exposed to, and estimating the number likely to be exposed.

Exposure concentration: The concentration of a substance in a given environment or environmental medium to which a person or organisms are exposed.

Exposure pathways: The ways in which people or other species are exposed to radiation or other contaminants. The three basic exposure pathways are ingestion, inhalation, and direct (external) exposures.

Gamma radiation: High-energy electromagnetic radiation emitted from various radionuclides when they transform from a higher energy to a lower energy state. The greatest penetrating power, but least ionizing, of the three principal forms of radiation from radioactive materials. Gamma radiation can completely penetrate and damage all body organs. Gamma radiation can be shielded effectively by several inches of lead, steel, concrete, or soil depending on the energy and intensity of the gamma radiation.

Grab sample: A single sample taken for analysis purposes either in a random manner or to target a suspected contaminated area.

Gray or Gy: Standard international unit for absorbed radiation dose, equal to the absorption of one joule of radiation energy per kilogram of material. Absorbed doses are frequently expressed in milligray (mGy), equal to one-thousandth of a gray, and must specify the medium in which the energy is absorbed.

Gross activity: The total response of a given detection system to an unknown mixture of radionuclides. Gross alpha (α), beta (β), or gamma (γ) measurements must be interpreted with caution and they are best used for screening or for following trends.

Groundshine: Radiation from contaminated soil and shoreline sediments.

Half-life: The time in which half the atoms of a particular radioactive substance disintegrate to another nuclear form. Each radionuclide has a unique half-life. Half-lives vary from millionths of a second to billions of years.

Hazard: Potential for radiation to cause human illness or injury. Hazard identification of a given substance is an informed judgment based on verifiable toxicity data from animal models or human studies.

Incremental: An increase, “in addition to.”

Internal dose: In a chemical exposure assessment, the amount of a substance penetrating the absorption barriers (e.g. skin, lung tissue, gastrointestinal tract) of an organism through either physical or biological processes. In radiological assessment, the radiation dose from radiation sources inside the body (see **absorbed dose**).

Ionizing radiation: Any radiation that disassociates electrons from atoms or molecules, thereby producing ions. Examples are alpha particles, beta particles, X-rays, and gamma rays.

Isotope: Differing forms of a particular natural element. The atoms of all forms will have the same number of protons in each nucleus and the same number of electrons surrounding the nucleus. Hence, the chemical behaviour of all forms will be essentially identical. However, each isotope’s nucleus will have a number of neutrons that is different from any other version. Thus, the isotopes (forms) of a particular element will have different physical properties, including the mass of its atoms and whether the nuclear structure of its atoms will retain its identity indefinitely (be “stable”) or undergo spontaneous

transformation (disintegration) at some future time (be "radioactive").

Mitigation: An action or design intended to reduce the severity or extent of an environmental impact.

Modelling: Using mathematical principles, environmental phenomenon are reported as an equation or series of equations (often as a computer program) to predict the change in some factor or variable as a result of changes in the other factors or variables in the equation.

Natural radioactivity: The property of radioactivity exhibited by more than 50 naturally occurring radionuclides.

Non-ionizing radiation: Radiation that is not capable of dislodging electrons from atoms or molecules (see **ionizing radiation**). Examples of non-ionizing radiation are radio waves, microwaves, and light.

Nucleus: The small positively charged core of an atom. It is only about 1/10,000th the diameter of the atom, but contains nearly all the atom's mass. All nuclei contain both protons and neutrons, except the nucleus of ordinary hydrogen; it consists of a single proton.

Nuclide: An atomic nucleus that contains a specific number of protons and neutrons. The nuclei of all isotopes of a given element have the same numbers of protons but different numbers of neutrons, and therefore are different nuclides.

Order of magnitude: A range of values between a specified lower value and an upper value 10 times as large.

Pathway: The physical course a radionuclide takes from its source to the exposed organism.

Pathways analysis: A method of estimating the transfer of contaminants (e.g. radionuclides released in water) and subsequently accumulated up the food chain to fish, vegetation, mammals, and humans, and the resulting radiological dose to humans.

Permafrost: Thermal conditions in the ground remaining below 0ºC continuously for more than 1 year.

Piezometer: A small slotted standpipe, usually hand-driven into the ground, used to measure water pressure, seepage of groundwater, and groundwater movement. It also can be used to sample near-surface groundwater.

Preliminary assessment: The process of collecting and reviewing available information about a known or suspected waste site or release.

Radiation: The emission and propagation of energy through space or matter in the form of electromagnetic waves (e.g. gamma rays) or fast-moving particles, such as alpha and beta particles.

Radioactive: The condition of a material exhibiting the spontaneous decay of an unstable atomic nucleus into a stable or unstable nucleus (e.g. U-238 decays into Th-234 (unstable) and Po-210 decays into Pb-206 (stable)).

Radionuclide: An element or isotope, which is radioactive as a result of the instability of the nucleus of its atom (e.g. Ra-226 or U-238).

Radon: A radioactive element in the U-238 decay chain produced by the radioactive decay of Ra-226. Radon is an inert gas. The half-life of Rn-222 is 3.8 days. Short-lived radon decay products or daughters (or progeny) are often the principal radiation hazards in underground mines. Decay of Rn-222 and short-lived daughters ultimately produces Pb-210.

Receptor: In human radiological assessments, a human being who is exposed to radioactivity released to the environment.

Reclamation: Restoration of a site to a beneficial use that may be for purposes other than the original use.

Relative biological effectiveness (RBE): A factor that reflects how different types of radiation differ quantitatively in producing biological effects. For example, if one radiation type requires 10 Gy to produce a biological effect and another type requires 5 Gy for the same effect, then the RBE of the second relative to the first is 10 Gy/5 Gy = 2.

Risk: An estimate of the probability that damage to life, health, property, and/or the environment will occur as a result of a given hazard.

Risk assessment: The process of qualitatively and/or quantitatively evaluating the potential risk posed to human health and/or the environment by the presence of specific pollutants.

Risk characterization: The last phase of the risk assessment process that estimates the potential probability for adverse health or ecological effects to occur from exposure to a stressor and evaluates the uncertainty involved.

Screening: A preliminary stage of the assessment process for rapid evaluation of relatively simple and routine activities or for determining the level of effort required for evaluating more complex projects.

Sievert or Sv: A unit of equivalent or effective dose used as a measure of the impact of ionizing radiation on humans. In theory, the unit Sv should be applied only at low doses and low-dose rates. Equivalent and effective doses are frequently expressed as millisievert (mSv), equal to one-thousandth of a

sievert, or as microsievert (μSv), equal to one-millionth of a sievert.

Slag: The residue of material separated during the refining and smelting of the pure mineral or desired mineral form.

Tailings: Residue of raw material separated out during the processing of raw mineral ores prior to smelting or refinement.

Uncertainty: A quantitative or qualitative expression of the robustness or reliability of a measured or calculated quantity as an accurate representation of the true value of that quantity.

Waste rock: That rock or mineral that must be removed from a mine in order to reach and obtain the minerals of interest, but it has no economic value.

APPENDIX A GENERIC FACTORS TO BE CONSIDERED IN RISK ASSESSMENT

For people residing in areas where local foods are important (e.g. berries, caribou, fish), it is important to consider the exposure from these food items. Where practicable, it is preferable to use measured site-specific data; however, these are not always available and it may be necessary to estimate the concentration in various environmental media. This appendix contains generic information that may be used to estimate radionuclide levels in various environmental media. The equations were provided in Section 4.0 of the main report. The information includes generic transfer factors for terrestrial vegetation, aquatic biota, and wildlife. To calculate the concentration in wildlife, it is necessary to multiply the transfer factor by an estimate of the intake. Generic values are also provided in this appendix to allow an estimate of the intake of some key wildlife that may be consumed by people.

It is important for users of this manual to understand that the parameters and factors presented in this appendix and elsewhere in the manual are provided to assist the assessor in situations where site-specific data are not available. It also must be understood that no warranty is provided as to the appropriateness of the data for any specific analysis; it is the responsibility of the assessor to ensure that the data used in any assessment are in fact appropriate for the site and the intended applications.

A1.0 ESTIMATING AIR CONCENTRATIONS OF RADIOACTIVE GASES AND PARTICLES

A useful model that has been used by many assessors to estimate radon concentrations in air near area sources is the Schiager Box Model, first proposed in 1974 (Schiager, 1974). Radioactive gases such as radon and thoron released from a large area source (tailings pile, field, etc.) will be subject to dilution in air and be dispersed by the prevailing wind. Where the release rate is uniform over the surface, a conservative estimate of the average radioactivity concentration in air at the edge of the area source can be calculated using the Schiager Box Model (Figure A1). The same general approach is often used to predict the emission of volatile chemicals (benzene, Hg^0, etc.) from the ground surface to outdoor air.

(A.1)

$$C_x = F\,L\,/\,(U\,\sigma_Z)\ \ \mathrm{Bq/m^3}$$

Where

C_x = average radionuclide concentration in air at the edge of an area source of width w and length L, Bq/m^3

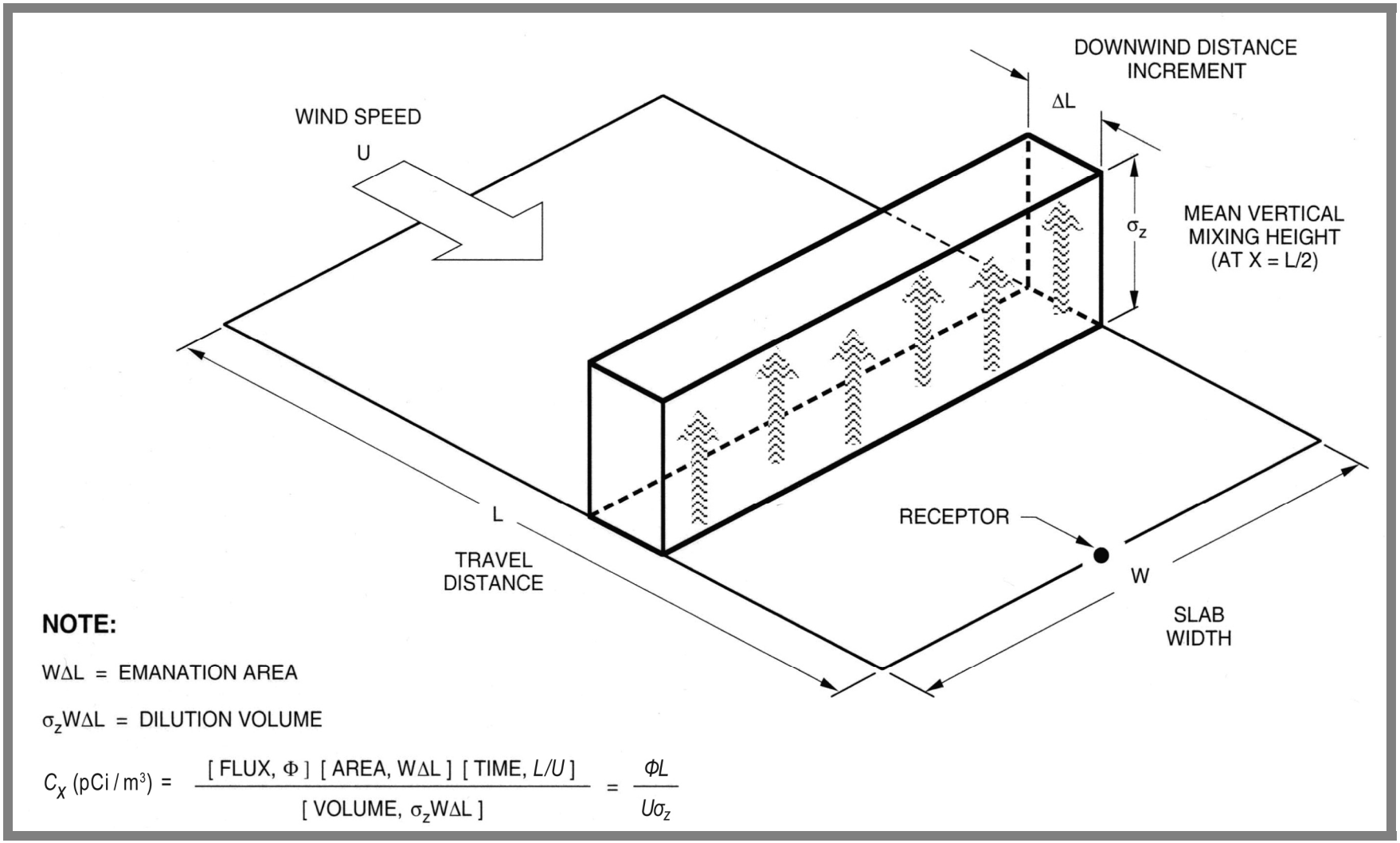
F = average emanation rate of radon from the surface, $Bq/(s\ m^2)$ (default = 1 $Bq/(m^2/s)$ per Bq Ra-226/g soil)

L = length of the emanating surface area, m

U = average wind speed, m/s

σ_Z = mean vertical dispersion coefficient at a distance of $L/2$, m

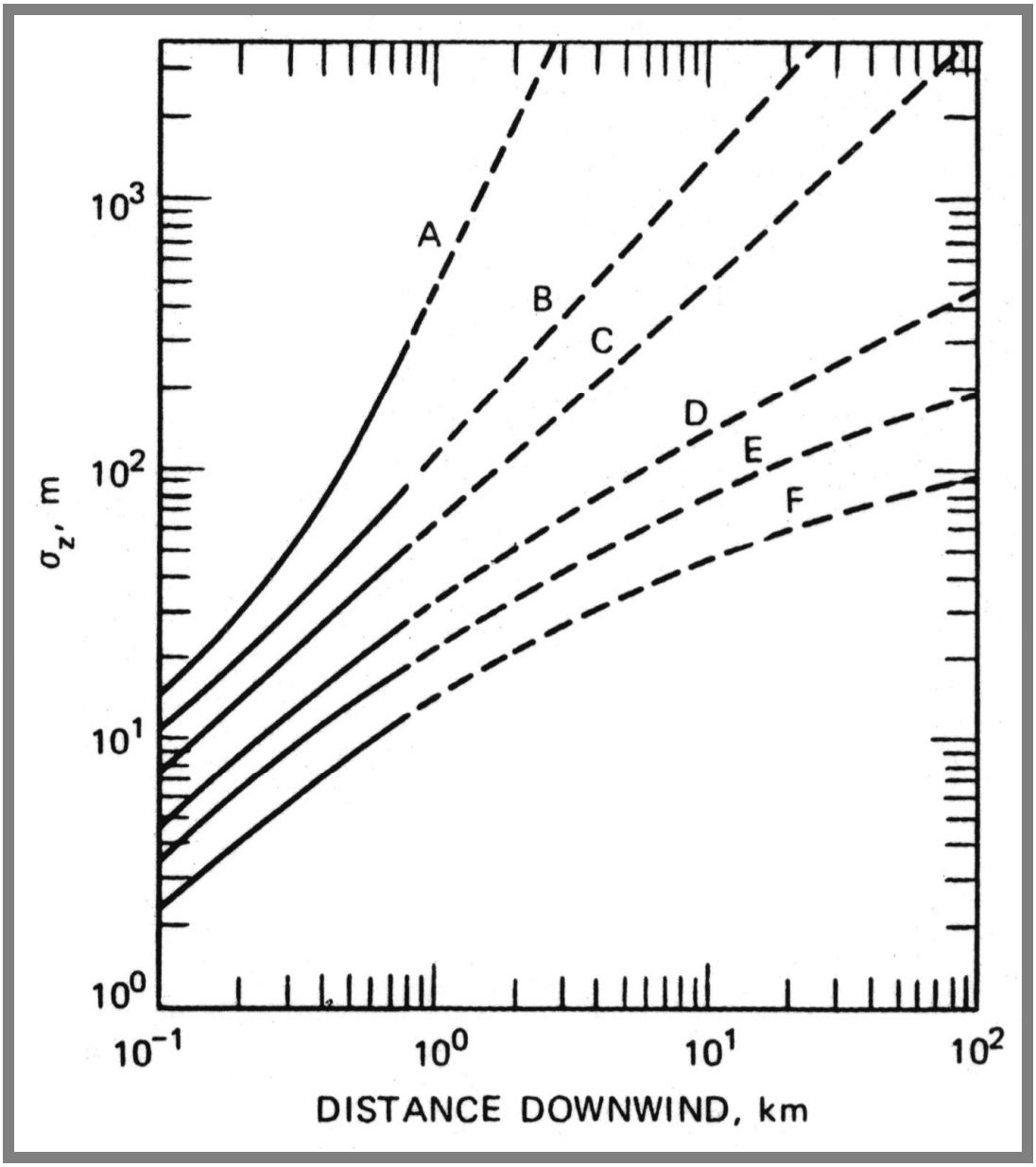
Figure A1. Schematic of Box Model for Concentration Calculations



Source: Schiager, 1974

The Canadian climate normals for site-specific, annual average wind speed values are recommended for use in risk assessment: the information can be obtained from the National Climate Data and Information Archives at www.canadainfolink.ca/climate.htm. The default values for σ_Z are provided in Figure A2.

Figure A2. Default Values of σ_Z



Source: Hanna et al., 1982.

A2.0 OTHER GENERIC PARAMETERS THAT COULD BE CONSIDERED IN RISK ASSESSMENT

In the absence of site-specific data for use at a site, it may be necessary to consider generic default factors in a screening-level assessment. This section provides a summary of some of the factors that could be considered in the assessment. These factors are used in the equations provided in Section 4.0 of the main report.

Climate data should be based on the specific site, and can be obtained from the Canadian climate normals data, National Climate Data and Information Archives, at www.canadainfolink.ca/climate.htm.

Table A1. Deposition Velocity

Particle size	Deposition Velocity (m/s)	Reference
< 1 µm	0.003	CSA, 1987

Table A2. Half-Lives and Decay Constants for Uranium Series Radionuclides

	Half-Life (y)	Decay Constant (y^{-1})
U-238	4.47×10^{9}	1.55×10^{-10}
Th-230	7.7×10^{4}	9.0×10^{-6}
Ra-226	1.6×10^{3}	4.33×10^{-4}
Pb-210	22	3.2×10^{-2}
Po-210	0.38 (138 d)	1.83

Note: y, year.
Source: ICRP, 1983.

Table A3. Yield Density for Different Types of Vegetation

Crop	Yield Density (kg/m^2)
Forage grass	0.28 (wet wt.)
Above-ground leafy vegetables	1.9 (wet wt.)
Non-leafy vegetables	0.57 (wet wt.)
Root vegetables	2.6 (wet wt.)
Fruits	0.31 (wet wt.)

Source: CSA,1987.

Table A4. Shore Width Factors for Shoreline Exposure

Exposure Location	Shore Width Factor
River Shoreline	0.2
Lake Shore	0.3
Nominal Ocean Site	0.5
Tidal Basin	1

A3.0 TRANSFER FACTORS

In the event that measured data are not available, it may be necessary to develop media concentrations for a given site depending on the conceptual model. These media concentrations are generally calculated using transfer factors. The use of transfer factors to predict concentrations in environmental media is a generally accepted practice in both chemical and radiological risk assessments.

Transfer factors that are used in the assessment should be selected based on the following process:

1. If site-specific transfer factors are available, these should be used in the assessment.
2. If site-specific data are inadequate or unavailable, regional specific transfer factors (e.g. transfer factors for Northwest Territories as opposed to a specific site) should be used if available.
3. If site-specific or regional specific transfer factors are not available, generic factors from the literature should be used.

The following tables provide generic transfer factors that can be considered for use in the equations that have been presented in Section 4.0. Deterministic values have been presented, and the sources of these values have been noted. It should be highlighted that the transfer factors provided in these tables are derived from literature sources using the same as those for chemicals; some of the transfer factors used for radionuclides are the same ones that are used for chemicals. An example of this is transfer factors for lead and lead-210; they are exactly the same because they behave the same way in the environment.

Transfer factors are a method of simplifying the complex process of uptake by different biota. Overall values are provided in this appendix that include complex exposure scenarios, e.g. a water-to-fish transfer factor that encompasses the uptake from ingestion of food and water, absorption, and exposure to sediment.
In the event that a probabilistic assessment becomes necessary, it is up to the risk assessor to select the most appropriate distributions to use. In general, the probabilistic assessment should use site-specific data. If site-specific data are not available, it is suggested that the risk assessor use log-normal distribution functions for the transfer factors with a geometric standard deviation of 2 to 3 and the minimum and maximum numbers being two standard deviations from the mean; this should encompass approximately 95% of the values.

Table A5. Generic Terrestrial Vegetation Transfer Factors

Parameter Description	Units	Default Values	Reference
Soil-to-plant transfer coefficient (B_v): summer forage			
U-nat	Bq/g (wet wt.) Bq/g (dry wt.)	1.8×10^{-2}	Létourneau, 1987; U.S. NCRP, 1996; IAEA, 1994
Th-230		9.2×10^{-3}	Létourneau, 1987; U.S. NCRP, 1996; IAEA, 1994
Pb-210		0.03	Létourneau, 1987; U.S. NCRP, 1996; U.S. EPA, 1998
Ra-226		0.093	Létourneau, 1987; U.S. NCRP, 1996; IAEA, 1994
Po-210		0.021	Létourneau, 1987; U.S. NCRP, 1996; IAEA, 1994
Soil-to-plant transfer coefficient (B_v): browse			
U-nat	Bq/g (wet wt.) Bq/g (dry wt.)	1.2×10^{-3}	Baes et al., 1984; IAEA, 1994; U.S. NCRP 1996
Th-230		1.4×10^{-4}	Baes et al., 1984; IAEA, 1994; U.S. NCRP 1996
Pb-210		5.0×10^{-3}	Baes et al., 1984; IAEA, 1994; U.S. NCRP, 1996; U.S. EPA, 1998
Ra-226		3.7×10^{-3}	Baes et al., 1984; IAEA, 1994; U.S. NCRP, 1996
Po-210		6.9×10^{-4}	Baes et al., 1984; U.S. NCRP, 1996
Soil-to-plant transfer coefficient (B_v): berry			
U-nat	Bq/g (wet wt.) Bq/g (dry wt.)	1.1×10^{-3}	Cassaday et al., 1985
Th-230		8.5×10^{-5}	Baes et al., 1984
Pb-210		9.0×10^{-2}	U.S. NRC, 1992
Ra-226		7.2×10^{-4}	Cassaday et al., 1985
Po-210		4.0×10^{-4}	Baes et al., 1984

Table A6. Generic Water-to-Sediment Distribution Coefficients

Parameter Description	Units	Default Value	Reference
U-nat	Bq/g (dry wt.) Bq/m^3	5.0×10^{-5}	IAEA, 1994; Bechtel Jacobs, 1998
Th-230		1.0×10^{-2}	IAEA, 1994; Bechtel Jacobs, 1998
Pb-210		9.0×10^{-4}	U.S. EPA, 1998
Ra-226		5.0×10^{-4}	IAEA, 1994; Bechtel Jacobs, 1998
Po-210		1.5×10^{-4}	Bechtel Jacobs, 1998

Table A7. Freshwater-to-Fish Transfer Factors

Parameter Description	Units	Default Value	Reference
U-nat	Bq/g (wet wt.) Bq/m^3	2.0×10^{-5}	CSA, 1987
Th-230		1.0×10^{-4}	IAEA, 1994; U.S. NCRP, 1996
Pb-210		3.0×10^{-4}	IAEA, 1994; U.S. NCRP, 1996
Ra-226		5.0×10^{-5}	IAEA, 1994; U.S. NCRP, 1996
Po-210		5.0×10^{-5}	IAEA, 1994

Table A8. Generic Water-to-Aquatic Vegetation Transfer Factors

Parameter Description	Units	Default Value	Reference
U-nat	Bq/g (wet wt.) Bq/m^3	2.0×10^{-4}	Santschi and Honeyman, 1989; ORNL, 1976; Bird and Schwartz, 1996; Létourneau, 1987
Th-230		2.6×10^{-3}	Santschi and Honeyman, 1989; ORNL, 1976; Bird and Schwartz, 1996; Létourneau, 1987
Pb-210		3.2×10^{-4}	Santschi and Honeyman, 1989; ORNL, 1976
Ra-226		9.7×10^{-4}	Santschi and Honeyman, 1989; ORNL, 1976; Bird and Schwartz, 1996; Létourneau, 1987
Po-210		1.8×10^{-3}	Santschi and Honeyman, 1989; ORNL, 1976

Table A9. Generic Water-to-Benthic Invertebrates Transfer Factors

Parameter Description	Units	Default Value	Reference
U-nat	Bq/g (wet wt.) Bq/m^3	1.0×10^{-4}	U.S. EPA, 1979
Th-230		5.0×10^{-4}	U.S. EPA, 1979; Létourneau, 1987
Pb-210		1.0×10^{-4}	U.S. EPA, 1979
Ra-226		2.5×10^{-4}	U.S. EPA, 1979
Po-210		2.0×10^{-2}	U.S. EPA, 1979

A4.0 CHARACTERISTICS OF SMALL AND LARGE GAME THAT COULD BE POTENTIALLY FOUND ON SITE

Characteristics of different large and small game that could potentially be found on FCSAP federally regulated contaminated sites can be identified from literature sources; as is the typical approach in risk assessments. Game that may be at the site, even seasonally, should be considered. References should be provided for all information, and detailed information should be provided regarding length of time spent at a site.

A5.0 WILDLIFE TRANSFER FACTORS

In general at contaminated sites, data to calculate transfer factors for wildlife are not available; therefore, wildlife transfer factors are generally derived from the literature as seen in the following tables. The transfer factors are multiplied by the intake of radionuclides (Bq/d) to obtain a wet weight flesh concentration (Bq/g). These transfer factors are generally derived from beef and poultry transfer factors. The sources of the transfer factors are provided in the tables. Because the sources of this data are well known, probability distribution functions have been derived from the data and are presented in these tables as well. If site-specific data are not available, these values can be used. For a probabilistic assessment, it is recommended that site-specific data are obtained to validate the modelled results. Rationale should be provided, where practicable, for use of transfer factors in risk assessment, including uncertainty and applicability.

Table A10. Generic Feed-to-Small Mammal (Hare)* Transfer Factors

Constituent	Units	Default Value	Parameter Distribution	Reference
U-nat	Bq/g (wet wt.) / Bq/d	3.5×10^{-5}	LN (3.5×10^{-5}, 3.0, 1.3×10^{-6}, 9.4×10^{-4})	Thomas, 1997†
Th-230		2.0×10^{-7}	LN (2.0×10^{-7}, 2.5, 1.3×10^{-8}, 3.1×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987§
Pb-210		1.4×10^{-4}	LN (1.4×10^{-4}, 3.0, 5.2×10^{-6}, 3.8×10^{-3})	Thomas, 1997†
Ra-226		2.5×10^{-3}	LN (2.5×10^{-3}, 3.0, 9.7×10^{-5}, 6.8×10^{-2})	Thomas, 1997†
Po-210		4.3×10^{-4}	LN (4.3×10^{-4}, 3.0, 1.6×10^{-5}, 1.2×10^{-2})	Thomas, 1997†

Note: The transfer factors are applied to the total intake by the receptor including food, water, and soil/sediment as appropriate. The intake rates of various food items are included in Section A4.0.

*Based in part on feed-to-beef transfer factors.
†Based on food chain concentration ratios for vegetation and voles.
§Based on feed-to-beef transfer factor information.

Table A11. Generic Feed-to-Moose* Transfer Factors

Constituent	Units	Default Value	Parameter Distribution	Reference
U-nat	Bq/g (wet wt.) / Bq/d	3.0×10^{-7}	LN (3.0×10^{-7}, 2.5, 1.9×10^{-8}, 4.7×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Th-230		2.0×10^{-7}	LN (2.0×10^{-7}, 2.5, 1.3×10^{-8}, 3.1×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Pb-210		4.0×10^{-7}	LN (4.0×10^{-7}, 2.5, 2.6×10^{-8}, 6.3×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Ra-226		1.0×10^{-7}	LN (1.0×10^{-7}, 2.5, 6.4×10^{-9}, 1.6×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Po-210		5.0×10^{-6}	LN (5.0×10^{-6}, 2.5, 3.2×10^{-7}, 7.8×10^{-5})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†

Note: The transfer factors are applied to the total intake by the receptor including food, water, and soil/sediment as appropriate. The intake rates of various food items are included in Section A4.0.

*Based on feed-to-beef transfer factors.
†Based on feed-to-beef transfer factor information.

Table A12. Generic Feed-to-Caribou* Transfer Factors

Constituent	Units	Default Value	Parameter Distribution	Reference
U-nat	Bq/g (wet wt.) / Bq/d	3.0×10^{-7}	LN (3.0×10^{-7}, 2.5, 1.9×10^{-8}, 4.7×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987§
Th-230		2.0×10^{-7}	LN (2.0×10^{-7}, 2.5, 1.3×10^{-8}, 3.1×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987§
Pb-210		1.0×10^{-6}	LN (1.0×10^{-6}, 2.0, 1.2×10^{-7}, 8.0×10^{-6})	Thomas et al., 1994†
Ra-226		1.0×10^{-7}	LN (1.0×10^{-7}, 2.5, 6.4×10^{-9}, 1.6×10^{-6})	IAEA, 1994; U.S. NCRP, 1996; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987§
Po-210		8.5×10^{-6}	LN (8.5×10^{-6}, 2.0, 1.2×10^{-6}, 6.8×10^{-5})	Thomas et al., 1994†

Note: The transfer factors are applied to the total intake by the receptor including food, water, and soil/sediment as appropriate. The intake rates of various food items are included in Section A4.0.

*Based mainly on feed-to-beef transfer factors.
†Calculated from lichen-to-caribou data for Pb-210 and Po-210.
§Based on feed-to-beef transfer factor information.

Table A13. Generic Feed-to-Bird* Transfer Factors

Constituent	Units	Default Value	Parameter Distribution	Reference
U-nat	Bq/g (wet wt.) / Bq/d	1.0×10^{-3}	LN (1.0×10^{-3}, 3.0, 3.7×10^{-5}, 2.7×10^{-2})	IAEA, 1994; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Th-230		1.0×10^{-4}	LN (1.0×10^{-4}, 3.0, 3.7×10^{-6}, 2.7×10^{-3})	IAEA, 1994; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Pb-210		2.0×10^{-4}	LN (2.0×10^{-4}, 3.0, 7.4×10^{-6}, 5.4×10^{-3})	IAEA, 1994; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†
Ra-226		3.0×10^{-4}	LN (3.0×10^{-4}, 3.0, 1.1×10^{-5}, 8.1×10^{-3})	Clulow et al., 1992§
Po-210		2.5×10^{-3}	LN (2.5×10^{-3}, 3.0, 9.3×10^{-5}, 6.8×10^{-2})	IAEA, 1994; Baes et al., 1984; U.S. EPA, 1998; CSA, 1987†

Note: The transfer factors are applied to the total intake by the receptor including food, water, and soil/sediment as appropriate. The intake rates of various food items are included in Section A4.0.

*Based on information for poultry. These transfer factors can be used for mallard, merganser, ptarmigan/grouse, and scaup.

†Based on feed-to-poultry information. When transfer factors were not available for poultry (Pb, Po, Th), the beef transfer factors were multiplied by a factor of 500 derived from the geometric mean of the ratio between the transfer factors for beef:chicken for Cd, Cu, Mo, Se, Zn, U, and Ra. A geometric standard deviation (GSD) of 3.0 was used with ± 3 standard deviations to span a range of three orders of magnitude (Baes et al., 1994). The ratios of transfer factors beef:poultry for Cd, Cu, Mo, Se, Zn, U, and Ra ranged up to a factor of 10 above and below the geometric mean of the ratios for the group. This variability and uncertainty was reflected in the predicted feed-to-poultry transfer factors for Pb, Po, and Th by expressing them as log-normal distributions that spanned three orders of magnitude.

§Default value for radium based on grouse. Based on a concentration ratio of 0.075fw and a feed ingestion rate of 224 g/d. A GSD of 3.0 was used with ± 3 standard deviations to span a range of three orders of magnitude (Baes et al., 1994).

Table A14. Generic Feed-to-Meat and Feed-to-Egg Transfer Factors

Radionuclide	Animal Product	Expected Transfer Factors (d/kg)	Range of Transfer Factors (d/kg)
Uranium	Beef	3×10^{-4}	-
Uranium	Pork	6.2×10^{-2}	-
Uranium	Poultry	1	0.3–1
Uranium	Egg contents	1	-
Thorium	-	-	-
Radium-226	Beef	9×10^{-4}	5×10^{-4}–5×10^{-3}
Lead-210	Beef	4×10^{-4}	1×10^{-4}–7×10^{-4}
Polonium-210	Beef	5×10^{-3}	6×10^{-4}–5×10^{-3}

Note: The transfer factors are applied to the total intake by the receptor including food, water, and soil/sediment as appropriate. The intake rates of various food items are included in Section A4.0.

Source: IAEA, 1994.

Table A15. Generic Feed-to-Cow Milk Transfer Factors

Radionuclide	Expected Transfer Factors (d/kg)	Range of Transfer Factors (d/kg)
Uranium	4×10^{4}	7.3×10^{-5}–6.1×10^{-4}
Thorium	-	-
Radium	1.3×10^{-3}	1×10^{-4}–1.3×10^{-3}
Lead	-	-
Polonium	3.4×10^{-4}	-

Source: IAEA, 1994.

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APPENDIX B RECOMMENDED DEFAULT RECEPTOR CHARACTERISTICS FOR USE IN RADIOLOGICAL/DOSE RISK ASSESSMENT

B1.0 DISCUSSION OF SELECTED RECEPTOR CHARACTERISTICS

Receptor characteristics, or "exposure factors," provide a quantitative description of the anatomical, physiological, and behavioural characteristics of an individual that have an impact on exposure to a stressor. The knowledge of such characteristics is critical for the estimation of exposure and risk. Some differences exist among the receptor characteristics that are typically used for chemical and radiological risk assessment because the characteristics have been provided by different agencies.

In Canada, radiological risk assessment has traditionally relied on receptor characteristics provided by the International Commission for Radiological Protection (ICRP), whereas chemical risk assessment has relied on variable receptor characteristics supplied by different governmental agencies at the provincial and federal levels. These receptor characteristics have typically been very different, with ICRP characteristics applying to individuals at a worldwide level and providing very detailed anatomical and physiological information. Chemical risk assessments have generally used summary data for Canadian populations. The ICRP receptor characteristics provide a lot of information not required to conduct chemical risk assessment, whereas receptor characteristics used in chemical risk assessment do not provide enough detail about the radiological dose that occurs following intake. Furthermore, the receptor age groups defined by the ICRP are different from those defined by most sources of Canadian and American receptor characteristics.

Despite these differences, harmonization of receptor characteristics used in radiological and chemical risk assessment is desirable and readily accomplished. The following discussion details how such harmonization can be achieved readily, mainly through the selection of common receptor characteristics for use in the calculations of both chemical and radiological risk assessment. The starting point of the receptor characteristics is the document developed by Richardson (1997), which is a statistical analysis of various receptor characteristics of the Canadian population.

The receptor characteristics relating to the anatomical, physiological, and behavioural characteristics of the whole organism can be the same in both radiological and chemical risk assessment. Characteristics relating to the anatomical and physiological characteristics of specific organ systems and those relating to the biokinetics of radionuclides are considered currently only in radiological risk assessment.

B1.1 International Commission on Radiological Protection Receptor Characteristics Versus Canadian Receptor Characteristics

Receptor characteristics from commonly used published Canadian sources were favoured for use over international receptor characteristics from the ICRP for several reasons. In its key Publication 23 (ICRP, 1975), the ICRP defines all anatomical and physiological values pertaining to "Reference Man," as well as listing the gross and elemental content of Reference Man. Reference Man was created to possess biological characteristics that were thought to be representative of all individuals worldwide, and was created for use in assessing radiological exposure and risk. The Reference Man suite also included a "Reference Woman" and a 10-year-old "Reference Child." Much of the data available for the Reference family are detailed physiological data at a smaller scale than is typically used in standard risk assessments but may be useful for other purposes, such as biokinetic modelling. Larger, whole body-scale characteristics that could be used included body weight, inhalation rate, and total fluid, tap water, and milk intake.

Several considerations were made in selecting receptor characteristics from Canadian sources over those presented by the ICRP (1975). These included:

- Non-dietary receptor characteristics presented by the ICRP include fewer age classes than the Canadian data. For the purposes of radiological risk assessment and selection of a critical group, receptor characteristics that are more specific to the group under examination are desirable.
- Receptor characteristics generated, based on primarily Canadian data, are likely to be more appropriate for use in risk assessment of Canadian populations compared with the generic international characteristics from the ICRP.
- The ICRP food intake rates are not age or gender specific, and are based on per caput (i.e. per person) rates that are not meant to estimate consumption rates for Reference Man. Food intake rates from Richardson (1997) are age and gender specific.
- The nutritional surveys presented by the ICRP were conducted approximately a decade before the Canadian nutritional surveys. The more recent data is likely to provide a better approximation of current food intake rates. Furthermore, it is likely that the studies that form the basis of other characteristics presented by the ICRP are older than those forming the basis of the Canadian data.

- The nutritional survey methods used to generate the food intake rates of the ICRP are likely less accurate than the 24-hour dietary recall methods used to generate the Canadian rates, and would introduce greater uncertainty as a result.
- Receptor characteristics of the ICRP do not provide probability distributions that would be required for probabilistic risk assessment. Canadian sources provide probability distributions for many receptor characteristics.

The preceding factors are considered sufficient for the recommendation of the use of Canadian receptor characteristics in this radiological risk assessment manual. However, if biokinetic modelling is necessary, the ICRP data may be more appropriate and these can be acquired from Publication 23 (ICRP, 1975).

B1.2 Application of Dose Conversion Factors

An additional element of radiological risk assessment is the conversion of external or internal dose into a measure of biologically effective dose by use of "dose conversion factors." The application of dose conversion factors in radiological risk assessment is a step that makes it different from chemical risk assessment. Only following application of dose conversion factors is it possible to compare dose estimates with regulatory dose limits or guidelines. Dose conversion factors are applied to the external or internal dose estimate to calculate the biological effects per unit of absorbed dose, because these differ depending on the type of radiation and the body part that is exposed (UNSCEAR 2000). Dose conversion factors incorporate information about age-specific physiological and anatomical features and make assumptions about age-specific absorbed radiation dose. The most widely accepted source for dose conversion factors is the ICRP. The age groups selected for receptor characteristics and those provided for the dose conversion factors by the ICRP are different, but can be combined if the age groups provided by the ICRP are selected.

The ICRP Publication 23 (ICRP, 1975) should be referred to if more detailed internal characteristics are required for radiological risk assessment, such as for models investigating the fate and transport of radionuclides throughout the body (e.g. biokinetic models).

B2.0 DESCRIPTION OF RECEPTOR CHARACTERISTICS SELECTED FOR USE IN THE RADIOLOGICAL RISK ASSESSMENT MANUAL

Data presented below is based on Health Canada PQRA (2004) guidance, and may be updated with more recent Health Canada publications, as relevant.

B2.1 Body Weight

Body weight probability distributions were taken from Richardson (1997). Richardson's values were based on a 1980 Nutrition Canada report (Demirjian, 1980) and from two Canadian Fitness and Lifestyle Research Institute reports produced in 1981 and 1988 (CFLRI, 1981, 1988).

B2.2 Soil Ingestion Rate

Soil ingestion rates for infants, toddlers, children, teens, and adults are from Health Canada (HC, 2004). These values were derived from Canadian Council of Ministers of the Environment (CCME) guidelines (CCME, 1996).

If no data on dust concentrations indoors are available, the concentrations of chemicals and radionuclides in dust are assumed to be the same as in soils. Few sources present dust intake rates as separate from soil intake rates. For this reason, several methods exist for deriving dust from soil intake rates. One method involves pro-rating the soil ingestion rate using the time spent indoors versus the time spent outdoors. This method is based on the assumption that receptors are exclusively exposed to dust while indoors and to soil while outdoors. To pro-rate the soil intake rate to derive the dust intake rate, it is multiplied by the ratio of time spent indoors in a day versus total time in a day. The ratio of time spent outdoors in a day versus total time in a day is multiplied by the soil intake rate to derive a more realistic soil intake rate.

B2.3 Inhalation Rate

Inhalation rate probability distributions for all ages were taken from Richardson (1997). These values were derived from probability density functions developed by Allan (1995) for Health Canada, using minute volumes, time-activity data, and a number of other factors. These probability distributions represent 24-hour inhalation rates.

B2.4 Water Ingestion Rate

Probability distributions for water ingestion rates were derived from Richardson (1997). Rates for the infant, toddler, child, and teen were based on an American 1977–78 tap

water intake study by Ershow and Cantor (1989), whereas rates for the adult were based on a North American 1977–78 tap water use study by the Environmental Health Directorate (EHD, 1981) and Health and Welfare Canada (unpublished) data cited by Richardson (1997).

B2.5 Time Spent Outdoors/Indoors

The amount of time spent outdoors and indoors was presented in time-activity data in Richardson (1997) that was averaged over 12 months for Canadians over the age of 11, and was thus used for teens and adults. The Richardson (1997) values were derived from a Statistics Canada (1993) cross-sectional population survey on individual time use. This survey was conducted using 24-hour recall diary methods and data were collected by telephone interview.

Time breakdowns for infants, toddlers, and children were derived from a yearly average based on the Canadian Human Activity Pattern Survey conducted in 1996–97 (Leech et al., 2002). Survey methods were similar to those in the Statistics Canada (SC, 1993) survey; the survey was conducted using 24-hour recall diary methods, and the interviews were conducted over the telephone.

B2.6 Skin Surface Area

Skin surface areas for all ages were taken from Richardson (1997), and were based on U.S. EPA (1995) equations that related total and partial skin surface area. Canadian data on height and body weight from the Nutrition Canada Survey (1970–72) (unpublished Health and Welfare Canada data cited by Richardson, 1997) were used in the U.S. EPA equations to predict skin surface areas for Canadians.

B2.7 Soil Adherence Factors

Soil adherence factors from Health Canada (2004), based on data from studies by Kissel et al. (1998, 1996), were adopted. These factors apply to hands, and to surfaces other than hands, but not to the total body. To derive factors for the full body, soil skin surface areas and their respective soil adherence factors (all from HC, 2004) were summed in a weighted fashion.

Available soil adherence data from Kissel et al. (1996, 1998) are not sufficient to provide a range of factors needed to perform a probabilistic assessment. For this reason, alternate soil adherence factors from U.S. EPA guidance (U.S. EPA 2001) are presented for use in probabilistic risk assessment should this be deemed necessary by the user. The U.S. EPA (2001) provides a range of soil adherence factors, from which the minimum and maximum factors based on human studies were selected. These factors apply to all parts of the body, whereas those of Kissel et al. (1996, 1998) are body-part specific. Additional information is available in U.S. EPA (2001), which provides activity-specific surface area weighted soil adherence factors, if needed.

B2.8 Human Milk Intake Rate

The human milk intake rate for infants less than 6 months of age were derived from Health Canada (HC, 1994) and were based on a 1983 Canadian study (NHW, 1983) that estimated Canadian human milk intake rates based on Swedish and American data. It was assumed that only infants consume human milk.

Infants were considered to either be either exclusively human milk-fed or non-human milk-fed. Human milk-fed infants were assumed to have a human milk intake and no water or baby formulae intake, whereas non-human milk-fed infants were assumed to have a water intake and a baby formulae intake but no human milk intake. These assumptions are based on a Health Canada assessment for priority substances (HC, 1994).

B2.9 Fraction of Vegetables and Fruit from Backyard Gardens

The Canadian Council of Ministers of the Environment (CCME, 1996) suggests that 10% of all fruit and vegetables are homegrown for residential situations and 50% for agricultural land use. If site-specific data are not available, these fractions should be applied to the intake data from Richardson (1997).

B2.10 Food Consumption Rates

Probability distributions for all food consumptions rates were based on Richardson (1997), which derived rates from data from Health and Welfare Canada (HWC, 1977). These reports published results of the Nutrition Canada Survey that was conducted from 1970–72. The survey used 24-hour recall survey methods.

Probability density functions for consumption rates of fruit and vegetables were based on rates in Richardson (1997) and Health and Welfare Canada (HWC, 1977). These rates included consumption of commercial and backyard garden fruit and vegetables. When consumption of backyard garden fruit and vegetables is included as an exposure pathway at a site, the user should calculate the amount of homegrown produce using site-specific values or the default values provided. Then the total amount of produce should be decreased to account for the homegrown produce. The arithmetic mean, geometric mean, minimum, and maximum values for these pathways need to be adjusted, whereas geometric standard deviation values remain the same.

The following tables provide receptor characteristics that should be used in both chemical and radiological risk assessments.

Table B1. Receptor Characteristics for Female Infant

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	8.2	7.8	1.4	4.0	15.3	Log-normal	Richardson, (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	2.1	2.01	1.32	1.15	3.50	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate†,‡ (L/d)	0.3	0.25	1.84	0.07	0.85	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 320 550 910 1,780						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.62×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				4×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Human milk consumption‡ (kg/d)	0.75						HC (1994) from NHW (1983)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Baby formulae‡	464	424	1.5	188	954	Log-normal	Richardson (1997) and HWC (1977)
Milk and dairy	664	602	1.6	235	1541		
Meat and eggs	52	42	1.9	11.6	152		
Fish and shellfish	-	-	-				
Root vegetables§	83	68	1.9	18.8	245		
Backyard root vegetables	6.05	4.96	1.9	1.37	17.9		
Other vegetables§	72	56	2.1	12.7	247		
Backyard other vegetables	5.25	4.08	2.1	0.93	18		
Fruits and juices§	120	97	1.9	26.9	350		
Backyard fruits and juices	3.49	2.82	1.9	0.78	10.2		
Cereals and grains	40	26	2.5	4.2	163		
Sugars and sweets	60	41	2.4	7.1	236		
Fats, nuts, and oils	26	19	2.2	3.9	92		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Infants are considered to either be either exclusively human milk-fed or non-human milk-fed. Human milk-fed infants have a human milk intake and no water or baby formulae intake, whereas non-human milk-fed infants have a water intake and a baby formulae intake but no human milk intake. These assumptions are based on HC (1994).

§Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

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Table B2. Receptor Characteristics for Male Infant

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum *	Maximum*	Distribution Type	Reference
Body weight (kg)	8.2	7.8	1.4	4.0	15.3	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	2.1	2.01	1.32	1.15	3.50	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate†,‡ (L/d)	0.3	0.25	1.84	0.07	0.85	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 320 550 910 1,780						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.62×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				4×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Human milk consumption‡ (kg/d)	0.75						HC (1994) from NHW (1983)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)		-				CCME (1996)
Food Consumption Rates (g/d)							
Baby	317	273	1.7	94	789	Log-normal	Richardson (1997)

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
formulae‡							and HWC (1977)
Milk and dairy	664	602	1.6	235	1541		
Meat and eggs	52	42	1.9	11.6	152		
Fish and shellfish	-	-	-				
Root vegetables§	83	68	1.9	18.8	245		
Backyard root vegetables	6.05	4.96	1.9	1.37	17.9		
Other vegetables§	72	56	2.1	12.7	247		
Backyard other vegetables	5.25	4.08	2.1	0.93	18		
Fruits and juices§	136	111	1.9	30.8	401		
Backyard fruits and juices	3.96	3.23	1.9	0.89	11.7		
Cereals and grains	40	26	2.5	4.2	163		
Sugars and sweets	60	41	2.4	7.1	236		
Fats, nuts, and oils	26	19	2.2	3.9	92		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Infants are considered to either be exclusively human milk-fed or non-human milk-fed. Human milk-fed infants have a human milk intake and no water or baby formulae intake, whereas non-human milk-fed infants have a water intake and a baby formulae intake but no human milk intake. These assumptions are based on HC (1994).

§ Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B3. Receptor Characteristics for Unisex Infant

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	8.2	7.8	1.4	3.98	15.29	Log-normal	Richardson, (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	2.1	2.01	1.32	1.15	3.50	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate†,‡ (L/d)	0.3	0.25	1.84	0.07	0.85	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 320 550 910 1,780						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.62×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				4×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Human milk consumption‡ (kg/d)	0.75						HC (1994) from NHW (1983)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Baby formulae‡	394	347	1.6	136	888	Log-normal	Richardson (1997) and HWC (1977)
Milk and dairy	664	602	1.6	235	1541		
Meat and eggs	52	42	1.9	11.6	152		
Fish and shellfish	-	-	-				
Root vegetables§	83	68	1.9	18.8	246		
Backyard root vegetables	6.05	4.96	1.9	1.37	17.9		
Other vegetables§	72	56	2.1	12.7	247		
Backyard other vegetables	5.25	4.08	2.1	0.93	18		
Fruits and juices§	136	111	1.9	30.8	401		
Backyard fruits and juices	3.96	3.23	1.9	0.89	11.7		
Cereals and grains	40	26	2.5	4.1	163		
Sugars and sweets	60	41	2.4	7.1	236		
Fats, nuts, and oils	26	19	2.2	3.9	92		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Infants are considered to either be exclusively human milk-fed or non-human milk-fed. Human milk-fed infants have a human milk intake and no water or baby formulae intake, whereas non-human milk-fed infants have a water intake and a baby formulae intake but no human milk intake. These assumptions are based on HC (1994).

§Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

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Table B4. Receptor Characteristics for Female Toddler

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	16.4	15.8	1.3	9.4	26.7	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.08						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	8.8	8.5	1.31	5.0	14.6	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	0.6	0.50	1.84	0.15	1.7	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 430 890 1,690 3,010						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.29×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				4×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Human milk consumption (kg/d)	-						HC (1994) from NHW (1983)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Baby formulae	495	428	1.7	148	1,237	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Milk and dairy	592	503	1.8	155	1,630		HWC (1977)
Meat and eggs	86	66	2.1	15	291		
Fish and shellfish	56	56	1.1	46	68		
Root vegetables‡	99	73	2.2	15	353		
Backyard root vegetables	7.22	5.32	2.2	1.1	26		
Other vegetables‡	67	45	2.4	7.8	259		
Backyard other vegetables	4.88	3.28	2.4	0.57	18.9		
Fruits and juices‡	234	181	2.0	45.25	724		
Backyard fruits and juices	6.81	5.27	2.0	1.32	21.1		
Cereals and grains	168	129	2.1	29.3	569		
Sugars and sweets	52	29	2.9	3.45	243		
Fats, nuts, and oils	24	16	2.4	2.78	92		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B5. Receptor Characteristics for Male Toddler

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	16.5	16.0	1.3	9.5	27.0	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.08						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	9.7	9.3	1.31	5.4	16.0	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	0.6	0.50	1.84	0.15	1.7	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 430 890 1,690 3,010						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.29×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				4×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Human milk consumption (kg/d)	-						HC (1994) from NHW (1983)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Baby formulae	495	428	1.7	148	1,237	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Milk and dairy	592	503	1.8	155	1,630		HWC (1977)
Meat and eggs	86	66	2.1	15	291		
Fish and shellfish	56	56	1.1	46	68		
Root vegetables‡	110	85	2.1	19.3	375		
Backyard root vegetables	8.02	6.2	2.1	1.41	27.3		
Other vegetables‡	67	45	2.4	7.8	259		
Backyard other vegetables	4.88	3.28	2.4	0.57	18.9		
Fruits and juices‡	234	183	2.0	45.8	732		
Backyard fruits and juices	6.81	5.33	2.0	1.33	21.3		
Cereals and grains	168	129	2.1	29.3	569		
Sugars and sweets	52	29	2.9	3.45	244		
Fats, nuts, and oils	28	19	2.4	3.3	109		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B6. Receptor Characteristics for Unisex Toddler

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	16.5	16.0	1.3	9.5	27.0	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.08						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	9.3	8.94	1.31	5.2	15.3	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	0.6	0.50	1.84	0.15	1.7	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Surface area weighted average	 430 890 1,690 3,010						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Total	 1×10^{-4} 1×10^{-5} 2.29×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				4×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Human milk consumption (kg/d)	-						HC (1994) from NHW (1983)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Baby formulae	495	428	1.7	148	1,237	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Milk and dairy	592	503	1.8	155.25	1,630		HWC (1977)
Meat and eggs	86	66	2.1	14.97	291		
Fish and shellfish	56	56	1.1	46	68		
Root vegetables‡	105	79	2.1	18	348		
Backyard root vegetables	7.65	5.76	2.1	1.31	25.4		
Other vegetables‡	67	45	2.4	7.8	259		
Backyard other vegetables	4.88	3.28	2.4	0.57	18.9		
Fruits and juices‡	234	183	2.0	45.8	732		
Backyard fruits and juices	6.81	5.33	2.0	1.33	21.3		
Cereals and grains	168	129	2.1	29.3	569		
Sugars and sweets	52	29	2.9	3.45	244		
Fats, nuts, and oils	26	17	2.5	2.72	106		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B7. Receptor Characteristics for Female Child

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*3	Maximum*	Distribution Type	Reference
Body weight (kg)	33.6	32.5	1.3	19.2	54.9	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	14.0	13.74	1.23	9.1	20.8	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	0.8	0.72	1.49	0.32	1.6	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 590 1,480 3,070 5,140						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.03×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic§				2×10^{-5}	1×10^{-4}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	579	488	1.8	151	1,581	Log-	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*3	Maximum*	Distribution Type	Reference
Meat and eggs	114	93	1.9	25.8	336	normal	HWC (1977)
Fish and shellfish	90	62	2.4	10.8	357		
Root vegetables‡	143	105	2.2	21.7	508		
Backyard root vegetables	10.42	7.65	2.2	1.58	37.1		
Other vegetables‡	98	65	2.5	10.4	406		
Backyard other vegetables	7.14	4.74	2.5	0.76	29.6		
Fruits and juices‡	252	185	2.2	38	895		
Backyard fruits and juices	7.33	5.38	2.2	1.11	26.1		
Cereals and grains	245	189	2.1	43	833		
Sugars and sweets	66	36	3.0	4	324		
Fats, nuts, and oils	40	27	2.4	4.7	156		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

§Values from U.S. EPA (2001) adjusted to reflect 2 years at the child adherence factor and 5 years at a lower rate to match age range defined by Health Canada.

Table B8. Receptor Characteristics for Male Child

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	32.2	31.2	1.3	18.5	52.7	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	15.1	14.73	1.25	9.43	23.0	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	0.8	0.72	1.49	0.32	1.6	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Total	 590 1,480 3,070 5,140						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 2.03×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic§				2×10^{-5}	1×10^{-4}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	645	539	1.8	166	1,746	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	131	103	2.0	25.8	412		HWC (1977)
Fish and shellfish	90	62	2.4	10.8	357		
Root vegetables‡	178	136	2.1	30.8	600		
Backyard root vegetables	12.98	9.91	2.1	2.25	44		
Other vegetables‡	98	65	2.5	10.4	406		
Backyard other vegetables	7.14	4.74	2.5	0.76	29.6		
Fruits and juices‡	283	217	2.1	49.2	957		
Backyard fruits and juices	8.24	6.31	2.1	1.43	27.9		
Cereals and grains	285	233	1.9	65	841		
Sugars and sweets	75	46	2.7	6.3	335		
Fats, nuts, and oils	49	33	2.4	5.7	190		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

§Values from U.S. EPA (2001) adjusted to reflect 2 years at the child adherence factor and 5 years at a lower rate to match age range defined by Health Canada.

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Table B9. Receptor Characteristics for Unisex Child

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body eight (kg)	32.9	31.8	1.3	18.8	53.7	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	14.5	14.15	1.25	9.1	22.1	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	0.8	0.72	1.49	0.32	1.6	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	118.5						Leech et al. (2002)
Skin surface area (cm^2) Hands Arms Legs Surface area weighted average	 590 1,480 3,070 5,140						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Total	 1×10^{-4} 1×10^{-5} 2.03×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic§				2×10^{-5}	1×10^{-4}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	613	513	1.8	158	1,662	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	123	98	2.0	24.5	392		HWC (1977)
Fish and shellfish	90	62	2.4	10.8	357		
Root vegetables‡	161)	119	2.2	24.6	576		
Backyard root vegetables	11.74	8.68	2.2	1.79	42		
Other vegetables‡	98	65	2.5	10.4	406		
Backyard other vegetables	7.14	4.74	2.5	0.76	29.6		
Fruits and juices‡	268	200	2.1	45.35	882		
Backyard fruits and juices	7.8	5.82	2.1	1.32	25.7		
Cereals and grains	265	211	2.0	53	844		
Sugars and sweets	71	41	2.9	4.9	345		
Fats, nuts, and oils	45	31	2.4	5.4	179		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

§Values from U.S. EPA (2001) adjusted to reflect 2 years at the child adherence factor and 5 years at a lower rate to match age range defined by Health Canada.

Table B10. Receptor Characteristics for Female Teen

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	56.2	55.1	1.2	38.3	79.3	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	14.0	13.74	1.22	9.2	20.5	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.0	0.86	1.73	0.29	2.6	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	90						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 800 2,230 4,970 8,000						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 1.90×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				1×10^{-5}	7×10^{-5}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	498	380	2.1	86	1,676	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	141	118	1.8	36.4	382		HWC (1977)
Fish and shellfish	104	78	2.1	17.7	344		
Root vegetables‡	190	151	2.0	37.8	604		
Backyard root vegetables	13.85	11.01	2.0	2.75	44.03		
Other vegetables‡	120	79	2.5	12.6	494		
Backyard other vegetables	8.75	5.76	2.5	0.92	36		
Fruits and juices‡	258	191	2.2	39.5	924		
Backyard fruits and juices	7.51	5.56	2.2	1.15	26.9		
Cereals and grains	232	172	2.2	35.5	832		
Sugars and sweets	66	41	2.6	6.1	277		
Fats, nuts, and oils	46	31	2.5	5.0	194		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B11. Receptor Characteristics for Male Teen

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	63.1	61.6	1.3	36.5	104	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	17.7	17.29	1.26	10.9	27.5	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.0	0.86	1.73	0.29	2.6	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	90						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 800 2,230 4,970 8,000						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 1.90×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				1×10^{-5}	7×10^{-5}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	674	518	2.1	117	2,284	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	201	166	1.9	46	599		HWC (1977)
Fish and shellfish	104	78	2.1	17.7	344		
Root vegetables‡	268	215	2.0	54	860		
Backyard root vegetables	19.54	15.67	2.0	3.9	63		
Other vegetables‡	120	79	2.5	12.6	494		
Backyard other vegetables	8.75	5.76	2.5	0.92	36		
Fruits and juices‡	258	191	2.2	39.5	924		
Backyard fruits and juices	7.51	5.56	2.2	1.15	26.9		
Cereals and grains	336	265	2.0	66	1,060		
Sugars and sweets	91	56	2.7	7.7	408		
Fats, nuts, and oils	71	45	2.6	6.7	304		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

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Table B12. Receptor Characteristics for Unisex Teen

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	59.7	58	1.2	40.3	83.5	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC(2004) from CCME (1996)
Inhalation rate (m^3/d)	15.8	15.33	1.28	9.4	25	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.0	0.86	1.73	0.29	2.6	Log-normal	Richardson (1997) from Ershow and Cantor (1989)
Time spent outdoors (min/d)	90						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 800 2,230 4,970 8,000						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 1.90×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				1×10^{-5}	7×10^{-5}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	583	441	2.1	100	1,945	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	170	138	1.9	38.2	498		HWC (1977)
Fish and shellfish	104	78	2.1	17.7	344		
Root vegetables‡	227	178	2.0	44.5	712		
Backyard root vegetables	16.55	12.98	2.0	3.24	51.9		
Other vegetables‡	120	79	2.5	12.6	494		
Backyard other vegetables	8.75	5.76	2.5	0.92	36.0		
Fruits and juices‡	258	191	2.2	39.5	924		
Backyard fruits and juices	7.51	5.56	2.2	1.15	26.9		
Cereals and grains	282	215	2.0	48.8	948		
Sugars and sweets	78	48	2.7	6.6	350		
Fats, nuts, and oils	58	37	2.6	5.5	250		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

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Table B13. Receptor Characteristics for Female Adult (20+ Years)

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	63.1	62.2	1.2	43.2	89.6	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	14.4	14.01	1.23	9.3	21.2	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.5	1.32	1.65	0.48	3.59	Log-normal	Richardson (1997) from EHD (1981) and unpublished Health and Welfare Canada work
Time spent outdoors (min/d)	90						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 890 2,500 5,720 9,110						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 1.88×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				1×10^{-5}	7×10^{-5}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	242	167	2.4	29	962	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	131	109	1.8	33.6	353		HWC (1977)
Fish and shellfish	104	72	2.4	12.5	415		
Root vegetables‡	157	123	2.0	30.8	492		
Backyard root vegetables	11.45	8.97	2.0	2.24	35.9		
Other vegetables‡	129	95	2.2	19.6	460		
Backyard other vegetables	9.4	6.93	2.2	1.43	33.5		
Fruits and juices‡	245	189	2.1	42.9	833		
Backyard fruits and juices	7.13	5.5	2.1	1.25	24.2		
Cereals and grains	181	138	2.1	31.3	609		
Sugars and sweets	56	37	2.5	5.9	231		
Fats, nuts, and oils	40	27	2.4	4.7	156		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B14. Receptor Characteristics for Male Adult (20+ Years)

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	78.8	77.5	1.2	53.82	111.6	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						Health Canada (2004) from CCME (1996)
Inhalation rate (m^3/d)	17.2	16.78	1.27	10.4	27.06	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.5	1.32	1.65	0.48	3.59	Log-normal	Richardson (1997) from EHD (1981) and unpublished Health and Welfare Canada work
Time spent outdoors (min/d)	90						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 890 2,500 5,720 9,110						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 1.88×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				1×10^{-5}	7×10^{-5}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	339	235	2.3	44.4	1,243	Log-normal	Richardson (1997) and HWC (1977)
Meat and eggs	209	171	1.9	47.37	617		

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Fish and shellfish	119	87	2.2	17.98	421		
Root vegetables‡	228 (211.38)	181 (167.81)	2.0	45	724		
Backyard root vegetables	16.62	13.19	2.0	3.3	52.8		
Other vegetables‡	148 (137.21)	107 (99.2)	2.2	22	517		
Backyard other vegetables	10.79	7.8	2.2	1.61	37.8		
Fruits and juices‡	245 (237.87)	189 (183.5)	2.1	43	833		
Backyard fruits and juices	7.13	5.5	2.1	1.25	24.25		
Cereals and grains	273	219	1.9	61	791		
Sugars and sweets	75	48	2.5	7.7	300		
Fats, nuts, and oils	59	42	2.3	7.9	222		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B15. Receptor Characteristics for Unisex Adult (20+ Years)

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	70.7	69.4	1.2	48.2	99.9	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.02						HC (2004) from CCME (1996)
Inhalation rate (m^3/d)	15.8	15.33	1.27	9.5	24.7	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.5	1.32	1.65	0.48	3.6	Log-normal	Richardson (1997) from EHD (1981) and unpublished Health and Welfare Canada work
Time spent outdoors (min/d)	90						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 890 2,500 5,720 9,110						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-4} 1×10^{-5} 1.88×10^{-5}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				1×10^{-5}	7×10^{-5}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Fraction of fruit from backyard garden (-)	0.1 (residential) 0.5 (agricultural)						CCME (1996)
Food Consumption Rates (g/d)							
Milk and dairy	286	196	2.4	34	1,129	Log-normal	Richardson (1997) and

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Meat and eggs	166	133	2.0	33	532		HWC (1977)
Fish and shellfish	111	79	2.3	14.9	418		
Root vegetables‡	188 (174.29)	147 (136.28)	2.0	36.8	588		
Backyard root vegetables	13.71	10.72	2.0	2.7	42.9		
Other vegetables‡	137 (127.01)	99 (91.78)	2.2	20.5	479		
Backyard other vegetables	9.99	7.22	2.2	1.49	34.9		
Fruits and juices‡	245 (237.87)	189 (183.5)	2.1	42	833		
Backyard fruits and juices	7.13	5.5	2.1	1.25	24.3		
Cereals and grains	222	172	2.1	39	759		
Sugars and sweets	65	42	2.5	6.7	263		
Fats, nuts, and oils	49	33	2.4	5.7	190		

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Consumption rates reflect total intake rates and include backyard fruits and vegetables as well as store-bought produce.

Table B16. Receptor Characteristics for Unisex Construction Worker (20+ Years)

Receptor Parameter	Arithmetic Mean	Geometric Mean	Geometric Standard Deviation	Minimum*	Maximum*	Distribution Type	Reference
Body weight (kg)	70.7	69.4	1.2	48.2	99.9	Log-normal	Richardson (1997) from Demirjian (1980) and CFLRI (1981, 1988)
Soil ingestion rate (g/d)	0.1						HC (2004) from MADEP (2002)
Inhalation rate (m^3/d)	15.8	15.33	1.27	9.5	24.7	Log-normal	Richardson (1997) and Allan (1995)
Water ingestion rate† (L/d)	1.5	1.32	1.65	0.48	3.6	Log-normal	Richardson (1997) from EHD (1981) and unpublished Health and Welfare Canada work
Time spent outdoors (min/d)	480						Richardson (1997) from Statistics Canada (1993)
Skin surface area (cm^2) Hands Arms Legs Total	 890 2,500 5,720 9,110						Richardson (1997) from unpublished Health and Welfare Canada work and U.S. EPA (1995)
Soil adherence factor (g/cm^2-event) Deterministic Hands Other Surface area weighted average	 1×10^{-3} 1×10^{-4} 1.88×10^{-4}						Based on HC (2004) from Kissel et al. (1996, 1998)
Soil adherence factor (g/cm^2-event) Probabilistic				2×10^{-5}	2×10^{-4}		U.S. EPA (2001)
Fraction of vegetables from backyard garden (-)	N/A						
Fraction of fruit from backyard garden (-)	N/A						
Food consumption rates (g/d)‡	N/A						

Note: N/A, this exposure pathway not applicable for the construction worker at the contaminated site.

*Minimum and maximum rates were calculated using two standard deviations of the geometric mean.

†Water intake rates include tap water as well as additional tap water-based beverages as per HC (2004).

‡Construction workers do not consume food from the contaminated site. However, depending on the assessment, the user may choose to include food consumption rates for the construction worker that are the same as the unisex adult.

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APPENDIX C EXAMPLE RADIOLOGICAL RISK ASSESSMENT FOR HYPOTHETICAL MINE SITE

C1.0 PROBLEM FORMULATION

C1.1 Objective

The objective is to determine whether there are any human health issues related to current exposures at a hypothetical abandoned mine site as a result of former activities at the site.

C1.2 Site Description

The area is characterized by its rocky and rugged relief with rock outcrops and sheer cliffs. Natural flat-lying land is, for the most part, non-existent in the area of the site or in the surrounding vicinity.

Soil cover in the vicinity of the site is generally sparse, and to the degree it exists, is generally very shallow. Within the immediate area of the site, sparse soil cover and vegetation are primarily in the shallow low-lying areas, on some of the hillside areas around the site, and in undisturbed areas. Bedrock generally predominates.

The area covered by waste rock or road fill is estimated to be in the order of 20 ha. For the most part, the steeper slopes are bare rock surfaces with no, or limited, vegetation cover. Only sparse vegetation of grasses, bushes, and pine trees covers much of the undisturbed areas of the site. The only edible plants at the site are berry plants. No medicinal plants, herbs, or other plants have been reported as being collected at the site.

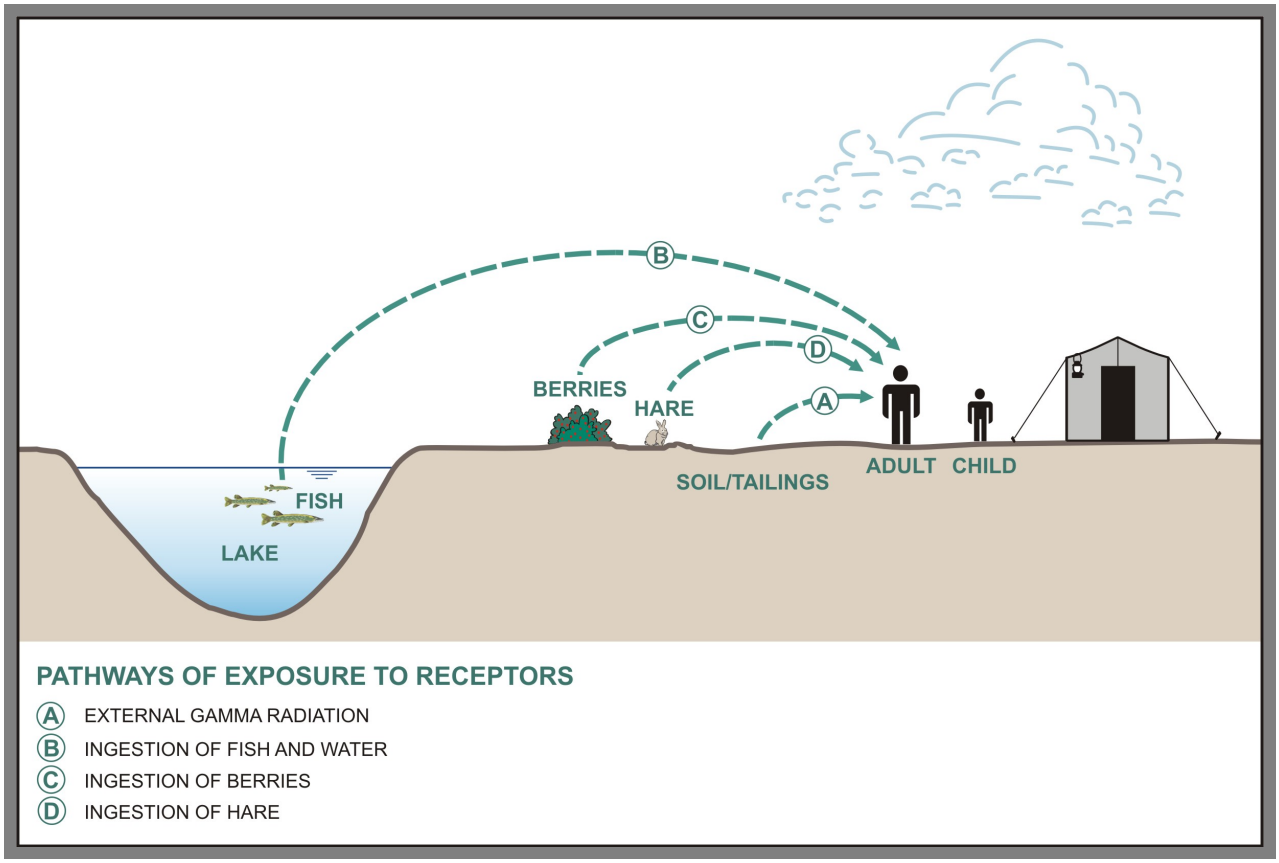
C1.3 Conceptual Model

The receptor group for consideration at this site is a camper who may have access to the site. The potential pathways of exposure include:

- external gamma
- ingestion of soil
- ingestion of water
- ingestion of berries
- ingestion of hare
- ingestion of fish

Figure C1 provides a schematic of the conceptual model for the hypothetical mine site.

Figure C1. Conceptual Model for the Site



The assessment is a screening-level risk assessment using maximum concentrations and conservative assumptions. In this assessment, only limited monitoring data were available, and it was considered appropriate to use maximum concentrations measured. In other assessments where more samples of environmental media are available, it may be appropriate to use other statistics, such as the arithmetic average of upper 95% confidence limit of the average, to describe the radionuclide concentrations in each medium.

C2.0 DATA COLLECTION AND EVALUATION

C2.1 Water Quality

Maximum measured radionuclide concentrations in water at the site are presented in Table C1.

Table C1. Maximum Measured Water Concentrations

Radionuclide	Maximum Water Concentration (Bq/L)
U-238	1.9
Th-230	<0.02*
Ra-226	0.02
Pb-210	0.02
Po-210	0.02
Th-228	0.02

*Measured as less than the detection limit, therefore half the detection limit was used.

Consistent with the recommended practice in radiological assessments where measured concentrations of radionuclides were below their respective detection limits, the assignment concentrations for purposes of dose calculations were set at half of the respective detection limits. This is also consistent with chemical risk assessment practice.

At the sample collection and analysis phase of this risk assessment, target minimum detection limits were developed for each radionuclide/environmental medium combination. The target detection limits were set to ensure that calculated doses could be determined at an acceptably small fraction of the dose criterion, and to ensure that there was optimum distribution of sample collection and analytical resources.

C2.2 Soil Quality

Maximum radionuclide levels from soil samples across the site are summarized in Table C2.

Table C2. Maximum Measured Soil Concentrations

Radionuclide	Maximum Concentration (Bq/g)
U-238	6.3
Th-230	3.8
Ra-226	8.4
Pb-210	4.3
Po-210	4.3
Th-228	0.06

C2.3 Terrestrial Vegetation Concentrations

Maximum radionuclide concentrations in berries measured on site are summarized in Table C3.

Table C3. Maximum Measured Berry Concentrations

Radionuclide	Maximum Concentration (Bq/g dry wt.)
U-238	0.011
Ra-226	0.031
Pb-210	0.02
Po-210	0.005

C2.4 Fish and Hare Concentrations

Fish and hare samples were collected from the site area. Table C4 provides a summary of the measured radionuclide concentrations in flesh (muscle). It is assumed that for purposes of this assessment, that individuals only consume the flesh (muscle) of the fish and hare.

Table C4. Measured Radionuclide Concentrations in Fish and Hare

Radionuclide	Hare (Bq/kg wet wt.)	Fish (Bq/kg wet wt.)
U-238	n/a	n/a
Th-230	n/a	n/a
Ra-226	<1.6	32.2
Pb-210	<3.1	<4.7
Po-210	1.2	0.8

Note: n/a, not available.

C2.5 External Gamma

The average gamma exposure rate measured at the site was 33 µR/h. The average dose rate was calculated over the accessible areas on the site where the group of receptors are expected to sleep, eat, and otherwise spend their time. The area over which the dose rate was averaged has no unique features that would affect the occupancy factor. Therefore, occupancy of all parts of the area was expected to be equally probable, barring specific reasons for a preference to a given location.

C2.6 Air Quality

Short-term air quality at the site was measured. The results of the assessment determined that the airborne concentrations of radionuclides including radon were not elevated above typical background levels.

The data indicated that inhalation exposures to airborne dust and radon on the site were likely similar to background levels, and thus were not considered further in this analysis.

C3.0 DOSE CHARACTERIZATION

The dose conversion factors for oral exposure used for this assessment are provided in Section 5.0 of this manual. As discussed in Section 6.0 of this manual, an essentially "negligible" dose limit of 0.05 mSv/year is being recommended at abandoned mine sites.

C4.0 EXPOSURE ASSESSMENT

As discussed previously, for the assessment it was assumed that a family would camp at the site for 3 months of the year. Therefore, an adult and a 6-year-old child were considered for the assessment of potential impacts to humans. The child was included because young children consume more food and water per unit body mass than adults, and therefore are more sensitive receptors. Toddlers were not included as human receptors in this assessment because the youngest children were not included in camping trips to this site. In many cases, toddlers would be left behind with an older family member, but they may also be considered in a risk assessment.

C4.1 Human Receptor Characteristics

The exposure to humans from radionuclides at this hypothetical mine site depends on behavioural characteristics, such as time at the site and source of drinking water. Conservative assumptions were made in the characterization of human receptors in this assessment.

For the purpose of the assessment, it was assumed that while at the site, the human receptors would obtain all their drinking water, berries, fish, and hare from local sources (i.e. relying on the site for their food and water). The limited size of the affected area was not expected to provide sufficient berries or game to be dried and carried back to home base for winter consumption.

The human receptor characteristics for the adult and child campers used for the assessment are shown in Table C5. Intake values were obtained from a food survey for an indigenous population in the area (Receveur et al., 1996).

Table C5. Human Receptor Characteristics for Campers

	Adult	Child
Fraction of year at site (-)*	0.25	0.25
Fraction of traditional food from local sources while at site (-)†	1.0	1.0
Fraction of water from site (-)†	1.0	1.0
Body weight (kg)‡	70.7	32.9
Hare ingestion rate (kg/d)§	0.002	0.001
Fish ingestion rate (kg/d)§	0.094	0.061
Berry ingestion rate (kg/d)§	0.0017	0.0011
Soil ingestion rate (g/d)‡	0.02	0.02
Water ingestion rate (L/d)‡	1.5	0.8
Berry ingestion rate (kg/d)§	0.0017	0.0011

*Seasonal, about 3 months of the year.

†Assumed source of traditional food and water.

‡ From characteristics provided in Appendix B.

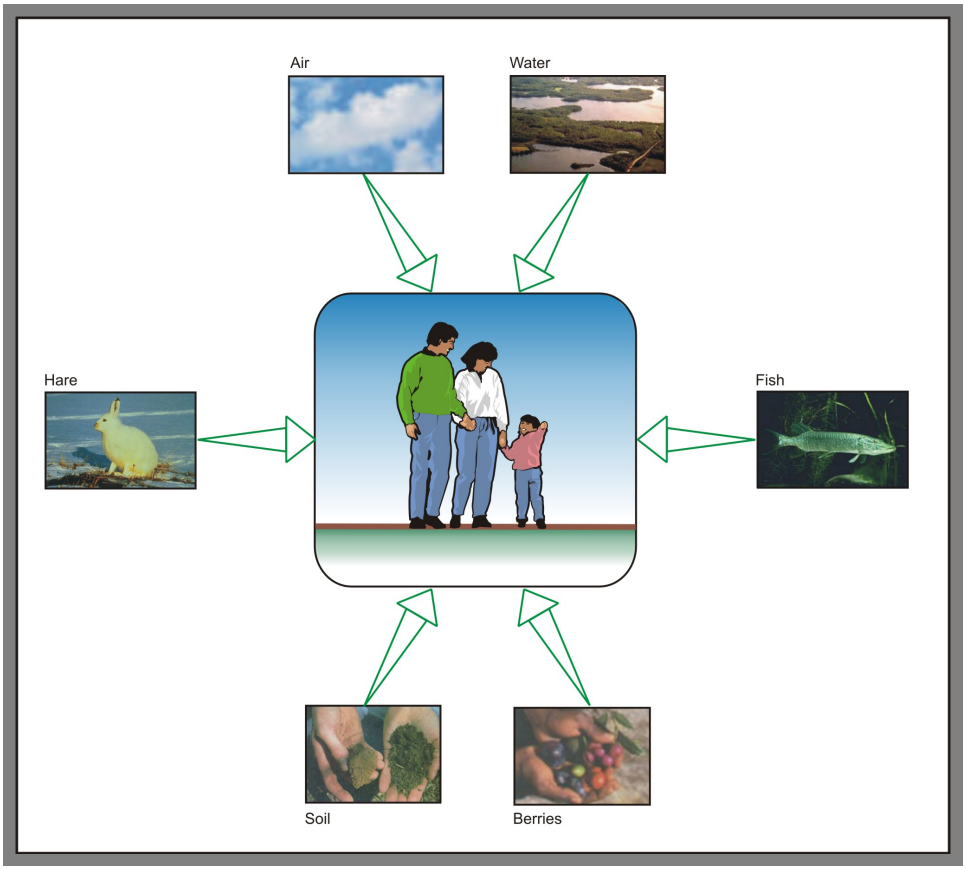
§Based on Receveur et al. (1996).

C4.2 Pathways of Exposure

The specific pathways of exposure considered at hypothetical mine site are shown in Figure C2 and include:

- consumption of drinking water from site at the maximum measured concentration
- consumption of fish flesh at the maximum measured concentrations
- consumption of hare flesh at the maximum measured concentration
- consumption of soil at the maximum measured concentration
- consumption of contaminated berries from the site
- exposure to external gamma radiation from contaminated soil and mine waste at the site.

Figure C2. Potential Pathways of Exposure for Humans



Potential exposures to radionuclides on site were calculated using the receptor characteristics and dietary information presented in Table C5. The measured concentrations in water (Table C1), soil (Table C2) berries (Table C3), fish (Table C4), and hare (Table C4) were used in the assessment. As there is no data available for some environmental media (e.g. U-238, Th-230, and Th-228 in fish), the concentrations were estimated following the equations provided in Section 4.0 and using the information in Appendix A. These calculations are shown in Table C6. Exposure to external gamma radiation based on data from the site was also included. The equations used to calculate exposures are presented in Section 4.0 of this manual.

Tables C7 and C8 provide the detailed calculations. As seen from these tables, the dose from exposure to external gamma at the site is the most significant pathway followed by the ingestion of fish from the site. The soil pathway is also important for the child exposure.

Table C6. Estimated Levels in Environmental Media

Fish

	Water Concentration Bq/L	Transfer Factor (Appendix A) Bq/g ww per Bq/m^3	Fish Concentration Bq/g ww
U-238	1.9	2.0E-05	3.8E-02
Th-230	0.01	1.0E-04	1.0E-03
Th-228	0.02	1.0E-04	2.0E-03

Fish concentration = water concentration (Bq/L) * TF (Bq/g ww per Bq/m3) * 1000 L/m^3

Berry

	Soil Concentration Bq/g	Transfer Factor (Appendix A) Bq/g ww per Bq/g	Berry Concentration Bq/g ww
Th-230	3.8	8.5E-05	3.2E-04
Th-228	0.06	8.5E-05	5.1E-06

Contribution from air is negligible for this site, therefore: Berry concentration = soil concentration (Bq/g) * TF (Bq/g ww per Bq/g)

Hare

Information on hare from Appendix A		
water intake	m^3/d	1.30E-04
total food intake (FW)	g (FW)/d	300
fraction that is browse	-	0.6
fraction that is forage	-	0.38
fraction that is soil	-	0.002
fraction of time in area	-	1

Forage

	Soil Concentration Bq/g	Transfer Factor (Appendix A) Bq/g ww per Bq/g	Forage Concentration Bq/g ww
U-238	6.3	1.8E-02	1.1E-01
Th-230	3.8	9.2E-03	3.5E-02
Th-228	0.06	9.2E-03	5.5E-04

Contribution from air is negligible for this site, therefore: Forage concentration = soil concentration (Bq/g) * TF (Bq/g ww per Bq/g)

Browse

	Soil Concentration Bq/g	Transfer Factor (Appendix A) Bq/g ww per Bq/g	Browse Concentration Bq/g ww
U-238	6.3	1.2E-03	7.6E-03
Th-230	3.8	1.4E-04	5.3E-04
Th-228	0.06	1.4E-04	8.4E-06

Contribution from air is negligible for this site, therefore: Forage concentration = soil concentration (Bq/g) * TF (Bq/g ww per Bq/g)

Intake

	Intake of Browse Bq/d	Intake of Forage Bq/d	Intake of Soil Bq/d	Intake of Water Bq/d	Total Intake Bq/d
U-238	1.4E+00	1.3E+01	3.8E+00	2.5E-01	1.8E+01
Th-230	9.6E-02	4.0E+00	2.3E+00	1.3E-03	6.4E+00
Th-228	1.5E-03	6.3E-02	3.6E-02	2.6E-03	1.0E-01

Intake = food ingestion rate (g/d) * fraction of food item * fraction of time at site * concentration (Bq/g)

Concentration

	Total Intake Bq/d	Transfer Factor (Appendix A) Bq/g ww per Bq/d	Hare Concentration Bq/g ww
U-238	1.8E+01	3.5E-05	6.4E-04
Th-230	6.4E+00	2.0E-07	1.3E-06
Th-228	1.0E-01	2.0E-07	2.1E-08

Concentration = intake (Bq/d) * TF (Bq/g per Bq/d)

Note: ww, wet wt.; TF, transfer factor

Table C7. Exposure and Dose Assessment for Adult Camper

Adult		**U-238+**	**Th-230**	**Ra-226+**	**Pb-210+**	**Po-210+**	**Th-228**	
Ingestion - Hare								
Hare conc	Bq/g ww	6.4E-04	1.3E-06	8.0E-04	1.6E-03	1.2E-03	2.1E-08	
DCF for ingestion	µSv/Bq	0.1	0.21	0.28	0.69	1.2	0.21	
Dose from ingestion	µSv/y	1.2E-02	4.9E-05	4.1E-02	2.0E-01	2.6E-01	7.9E-07	5.1E-01
= hare conc*oral DCF adult (Section 5) * adult hare ingestion rate (Table D5) * 1000 (conversion factor g/kg) * 365d/y * fraction of food from site (Table D5) * fraction of time at site (Table D.5)								
Ingestion - Soil								
Soil conc	Bq/g	6.3E+00	3.8E+00	8.4E+00	4.3E+00	4.3E+00	6.0E-02	
Dose from ingestion	µSv/y	1.1E+00	1.5E+00	4.3E+00	5.4E+00	9.4E+00	2.3E-02	2.2E+01
= soil conc * oral DCF adult (Section 5) * adult soil ingestion rate (Table D.5) * 365d/y * fraction of time at site (Table D5)								
Ingestion - Fish								
Fish conc	Bq/g ww	3.8E-02	1.0E-03	3.2E-02	2.4E-03	8.0E-04	2.0E-03	
Dose from ingestion	µSv/y	3.3E+01	1.8E+00	7.7E+01	1.4E+01	8.2E+00	3.6E+00	1.4E+02
= fish conc * oral DCF adult (Section 5) * adult fish ingestion rate (Table D5) * 1000 (conversion factor g/kg) * 365d/y * fraction of food from site (Table D5) * fraction of time at site (Table D5)								

Ingestion - Water

Water conc	Bq/L	1.9E+00	1.0E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02	
Dose from ingestion	µSv/y	2.6E+01	2.9E-01	7.7E-01	1.9E+00	3.3E+00	5.7E-01	3.3E+01

= water conc * oral DCF adult (Section 5) * adult water igestion rate (Table D5) ** 365d/y * fraction of time at site (Table D5)

Ingestion - Berry

Berry conc	Bq/g dw	1.1E-02	3.1E-02	2.0E-02	5.0E-03			
Berry conc	Bq/g ww*	3.3E-03	3.2E-04	9.3E-03	6.0E-03	1.5E-03	5.1E-06	
Dose from ingestion	µSv/y	5.1E-02	1.1E-02	4.0E-01	6.4E-01	2.8E-01	1.7E-04	1.4E+00

= berry conc * oral DCF adult (Section 5) * adult berry ingestion rate (Table D5)* 1000 (conversion factor g/kg) * 365d/y*fraction of food from site (Table D5) * fraction of time at site (Table D5)

External Gamma

External Gamma rate	µR/h	33
Dose conversion	µSv/µR	0.006
External dose	µSv/y	434

= external gamma rate * DCF for adult (Section 5) * 24 h/d * 365d/y * fraction of time at site (Table D5)

TOTAL **628 µSv/y**

Note: DCF, dose conversion factor; y, year; ww, wet wt.; TF, transfer factor; conc, concentration
* wet weight concentration estimated from a dry weight concentration assuming a 70% moisture content.

Table C8. Exposure and Dose Assessment for Child Camper

Child		U-238+	Th-230	Ra-226+	Pb-210+	Po-210+	Th-228	
Ingestion - Hare								
Hare conc	Bq/g ww	6.4E-04	1.3E-06	8.0E-04	1.6E-03	1.2E-03	2.1E-08	
DCF for ingestion	µSv/Bq	0.185	0.31	0.62	2.2	4.4	0.57	
Dose from ingestion	µSv/y	1.4E-02	4.8E-05	6.0E-02	4.1E-01	6.4E-01	1.4E-06	1.1E+00
= hare conc*oral DCF child (Section 5) * child Hare Ingestion Rate (Table D5) * 1000 (conversion factor g/kg) * 365d/y * fraction of food from site (Table D5)*fraction of time at site (Table D5)								
Ingestion - Soil								
Soil conc	Bq/g	6.3E+00	3.8E+00	8.4E+00	4.3E+00	4.3E+00	6.0E-02	
Dose from ingestion	µSv/y	2.1E+00	2.1E+00	9.5E+00	1.7E+01	3.5E+01	6.2E-02	6.6E+01
= soil conc * oral DCF child (Section 5) * child soil ingestion rate (Table D5) * 365d/y * fraction of time at site (Table D5)								
Ingestion - Fish								
Fish conc	Bq/g ww	3.8E-02	1.0E-03	3.2E-02	2.4E-03	8.0E-04	2.0E-03	
Dose from ingestion	µSv/y	3.9E+01	1.7E+00	1.1E+02	2.9E+01	2.0E+01	6.3E+00	2.1E+02
= fish conc * oral DCF child (Section 5)* child fish ingestion rate (Table D5) * 1000 (conversion factor g/kg) * 365d/y * fraction of food from site (Table D5) * fraction of time at site (Table D5)								
Ingestion - Water								
Water conc	Bq/L	1.9E+00	1.0E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02	
Dose from ingestion	µSv/y	2.6E+01	2.3E-01	9.1E-01	3.2E+00	6.4E+00	8.3E-01	3.7E+01
= water conc * oral DCF child (Section 5) * child water ingestion rate (Table D5) ** 365d/y * fraction of time at site (Table D5)								
Ingestion - Berry								
Berry conc	Bq/g dw	1.1E-02	3.1E-02	2.0E-02	5.0E-03			
Berry conc	Bq/g ww*	3.3E-03	3.2E-04	9.3E-03	6.0E-03	1.5E-03	5.1E-06	
Dose from ingestion	µSv/y	6.1E-02	1.0E-02	5.8E-01	1.3E+00	6.6E-01	2.9E-04	2.6E+00
= berry conc * oral DCF child (Section 5) * child berry ingestion rate (Table D5) * 1000 (conversion factor g/kg) * 365d/y * fraction of food from site (Table D5) * fraction of time at site (Table D5)								
External Gamma								
External Gamma rate	µR/h	33						
Dose conversion	µSv/µR	0.008						
External dose	µSv/y	578						
= external gamma rate * DCF for child (Section 5) * 24 h/d * 365d/y*fraction of time at site (Table D5)								
TOTAL	**892 µSv/y**							

Note: DCF, dose conversion factor; y, year; ww, wet wt.; TF, transfer factor; conc, concentration.
* wet weight concentration estimated from a dry weight concentration assuming a 70% moisture content

C5.0 DOSE ASSESSMENT

Assessment of radiation exposures to members of the public is commonly based on estimation of the incremental effects of the project or site. Such assessments consider the radiation dose received from direct exposure to gamma radiation, as well as the dose received from ingestion of radionuclides. As seen from the tables in this appendix, the predicted doses for both adult and children who would potentially camp at the site are above the Health Canada essentially "negligible" dose limit of 0.05 mSv/year, but well below the Canadian Nuclear Safety Commission dose limit of 1 mSv/year. The major portion of the dose comes from exposure to external gamma at the site. It should be noted that the dose from external gamma alone exceeds the 0.05 mSv/year "negligible" dose limit.

The results of this assessment indicate that further investigation is needed at this site.

C5.1 Next Steps

The custodian at the site, with input from the risk assessor, will determine the follow-up steps to this assessment. They can be as follows:

- Collect additional data.
- Refine the assumptions at the site and redo risk assessment with less conservative and more realistic assumptions. It was assumed that the campers would be exposed to the average gamma rate at the site. It may be prudent to go back to the communities that potentially use the site and determine how long they would be present at the site, what their uses are of the site, and the approximate locations that they would use at the site for establishing their camp. As indicated in the site description, the site does not have many flat areas where camps would be set up.
- The custodial department may decide to go ahead and develop a remedial plan for the site to reduce the exposure to external gamma. The option for reducing gamma radiation might include the placement of a layer of rock or till. The thickness and extent of the layer, and the availability and cost of transporting rock or till to the site would be considered in determining the remedial plan.

C6.0 REFERENCE

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