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Canadian Soil Quality Guidelines for Uranium: Environmental and Human Health

Scientific Supporting Document

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ABSTRACT

Canadian environmental quality guidelines, developed under the auspices of the Canadian Council of Ministers of the Environment (CCME), are numerical concentrations or narrative statements recommended to support and maintain designated resource uses. Canadian soil quality guidelines can be used as the basis for consistent assessment and remediation of contaminated sites in Canada.

This report was prepared by Health Canada and the National Guidelines and Standards Office of Environment Canada, which acts as Technical Secretariat for the CCME Soil Quality Guidelines Task Group. The Guidelines were derived according to the procedures described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 2005).

Following the introduction, Chapter 2 presents chemical and physical properties of uranium and a review of the sources and emissions in Canada. Chapter 3 discusses uranium's distribution and behavior in the environment. Chapter 4 discusses the behavioural effects and bioaccumulation of uranium in biota, including soil microbial processes, terrestrial plants, animals and livestock and wildlife. Chapter 5 reports toxicological effects of uranium in humans and mammalian species. The above information is reflected in Chapter 6 which outlines the derivation procedure for the calculation of soil quality guidelines for uranium to protect human and environmental receptors in four types of land uses: agricultural, residential/parkland, commercial, and industrial.

The following soil quality guidelines for uranium are recommended by CCME based on the available scientific data. The uranium environmental soil quality guideline (SQG_E) for agricultural land use is 33 mg/kg soil. An SQG_E of 500 mg/kg soil was determined for residential/parkland land use, and an SQG_E of 2000 mg/kg soil was determined for commercial and industrial land uses. The preliminary human health soil quality guidelines ($PSQG_{HH}$) for uranium are 23 mg/kg soil for agricultural land uses, 23 mg/kg soil for residential/parkland land uses, 33 mg/kg soil for commercial land use, and 300 mg/kg soil for industrial land use. Therefore, the final SQGs for uranium for the protection of both human and environmental health are 23 mg/kg soil for agricultural land use, 23 mg/kg soil for residential/parkland land use, 33 mg/kg soil for commercial land use and 300 mg/kg soil for industrial land use.

RÉSUMÉ

Les recommandations canadiennes pour la qualité de l'environnement, élaborées sous les auspices du Conseil canadien des ministres de l'environnement (CCME), sont des valeurs de concentrations ou des énoncés décrivant des conditions recommandées afin d'assurer le maintien et le développement durable d'utilisations désignées des ressources. On peut se fonder sur les *Recommandations canadiennes pour la qualité des sols* proposées par le CCME pour conformer l'évaluation et l'assainissement des lieux contaminés au Canada.

Le présent document a été préparé par Santé Canada et le Bureau national des recommandations et des normes d'Environnement Canada, qui fournit des services de secrétariat technique au groupe de travail du CCME sur les recommandations pour la qualité des sols. On a élaboré ces recommandations selon les procédures décrites dans le *Protocole d'élaboration de recommandations pour la qualité des sols en fonction de l'environnement et de la santé humaine* (CCME 2005).

Après une brève introduction, le chapitre 2 présente les propriétés chimiques et physiques de l'uranium, de même qu'un aperçu des sources et des émissions au Canada; le chapitre 3 traite de la distribution et du devenir de cette substance dans l'environnement; le chapitre 4 examine les effets et la bioaccumulation de l'uranium dans le biote, notamment sur les processus microbiens, les plantes terrestres, les animaux et le bétail; le chapitre 5 rapporte les effets toxicologiques de l'uranium sur les humains et d'autres espèces de mammifères, et ces informations, reprises au chapitre 6, servent à l'élaboration des recommandations pour la qualité des sols pour l'uranium en vue de protéger la santé humaine et l'environnement, selon quatre types d'utilisations des terrains (agricole, résidentielle/parcs, commerciales et industrielles).

Les valeurs ci-dessous des recommandations pour la qualité des sols pour l'uranium du CCME sont fondées sur les données scientifiques disponibles. La recommandation pour la qualité des sols visant l'uranium (RQS_E) (environnement) est de 33 mg/kg pour les terres agricoles. On a défini une RQS_E de 500 mg/kg de sol l'utilisation résidentielle et les parcs, et de 2000 mg/kg de sol pour les sols à vocation commerciale et industrielle. Les recommandations pour la qualité des sols (santé humaine) (RQS_{SH}) pour l'uranium sont de 23 mg/kg de sol à des fins d'utilisation agricole, 23 mg/kg de sol à des fins d'utilisation résidentielle/parc, 33 mg/kg pour les sols à vocation commerciales, et 300 mg/kg de sol à des fins industrielles. Donc, les RQS finales pour l'uranium (protection de la santé humaine et l'environnement) sont de 23 mg/kg de sol à des fins d'utilisation agricole, 23 mg/kg de sol pour l'utilisation résidentielle/parc, 33 mg/kg de sol pour les sols à vocation commerciale et 300 mg/kg de sol à des fins industrielles.

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CHAPTER 1. INTRODUCTION

Canadian Soil Quality Guidelines are numerical concentrations or narrative statements that specify levels of toxic substances or other parameters in soil that are recommended to maintain, improve or protect environmental quality and human health. They are developed using formal protocols to ensure nationally consistent, scientifically defensible values. The guidelines are nationally endorsed through the Canadian Council of Ministers of the Environment (CCME).

This report reviews the sources and emissions of uranium, its distribution and behaviour in the environment, and its non-radiological (chemical) toxicological effects on soil microorganisms, plants, animals, and humans. Guidelines are derived according to *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 2005) for various land uses: agricultural, residential/parkland, commercial and industrial. In addition, various check mechanisms considering indirect pathways of exposure (e.g., nutrient and energy cycling check and off-site migration of contaminants via wind and water erosion) are used to provide protection for resources and receptors not otherwise considered in the derivation of soil quality guidelines.

The following derived values should be considered for general guidance purposes; however, in the application of these values, site-specific conditions should be considered. Because the guidelines may be applied differently in various jurisdictions, the reader should consult appropriate authorities for guidance in the application of these guidelines. Every attempt was made to provide a conservative estimate that could be applied to any area in Canada. Due to geological conditions, it is possible that natural enrichment will result in exceedances of the soil quality guidelines. Thus, such exceedances do not automatically imply that the ecosystem is compromised. The guideline represents a limit below which no adverse impacts are expected, but site-specific information, such as local background concentrations, should always be considered in the application of these guidelines.

CHAPTER 2. BACKGROUND INFORMATION

Physical and Chemical Properties

Uranium, in its pure form is a silvery-white, weakly radioactive metal. It is the heaviest of the naturally occurring elements and is present in the earth's crust at an average concentration of 0.0003% (i.e., 3 parts per million or 3 mg/kg). Uranium is found in a variety of chemical forms in all soils, oceans, food, and drinking water (Bleise 2003). Uranium deposits have been reported across Canada, with several large mineral deposits containing uranium at concentrations greater than crustal abundance. The richest uranium deposit in the world is in northern Saskatchewan (Painter *et al. 19*94). Uranium occurs in 5 oxidation states (+2, +3, +4, +5, +6); however, only two oxidation states (+4, +6) are considered stable enough to be of practical importance (ATSDR 1999). The physical and chemical properties of uranium metal are summarized in Table 1.

In the earth's crust, uranium is generally found as oxides, such as uranium dioxide (UO_2) or triuranium octaoxide (U_3O_8). The mineral pitchblende, the main uranium ore, consists primarily of uranium oxides. In soil, uranium is primarily (80-90%) present in the +6 oxidation state as the uranyl cation (UO_2^{2+}) (Ebbs et al. 1998). Speciation of uranium in soil and aqueous systems is pH-dependent. Therefore, under acidic reducing conditions, UO_2^{2+} is the predominant uranium species in the soil; under neutral conditions, hydroxide complexes such as UO_2OH^+, $(UO_2)_2(OH)_2^{2+}$,$(UO_2)_3(OH)^{5+}$ and $(UO_2)_3(OH)_7^-$ and phosphate complexes such as $UO_2HPO_4^0$ and $UO_2(HPO_4)_2^{2-}$ form; under alkaline conditions, carbonate complexes such as $UO_2CO_3^0$, $UO_2(CO_3)_2^{2-}$ and $UO_2(CO_3)_3^{4-}$ predominate (Ebbs et al. 1998).

Naturally occurring isotopes of uranium include ^{234}U, ^{235}U, and ^{238}U (BEIR IV 1988; Merck Index 1989). The ^{238}U isotope is the most abundant by weight (99.28%) followed by substantially lower quantities of ^{235}U (0.72%) and ^{234}U (0.0055%). These three isotopes are each radioactive, which means that the nuclei of the atoms spontaneously disintegrate or "decay" as they transform into different, more stable atoms. This decay results in the emission of radioactivity consisting of alpha particles (subatomic fragments consisting of two protons and two neutrons), beta particles (fast-moving electrons ejected from the nuclei of atoms), and gamma rays (electromagnetic energy). Uranium continually undergoes transformation, releasing energy (alpha and beta particles and gamma rays) through a serial production of a chain of decay products (progeny) until a stable element has formed. The majority of natural uranium radioactivity (97.8%) is due to the isotopes ^{238}U (48.9%) and ^{234}U (48.9%). The parent isotope of the uranium series is ^{238}U (of which ^{234}U is a decay product) while ^{235}U is the parent isotope of the actinide series (ATSDR 1999). The occurrence, half-lives, and radioactive properties of uranium isotopes are summarized in Table 2. The decay products of these naturally occurring isotopes are provided in Table 3.

The ^{235}U isotope is of use in nuclear reactors as it is fissile and therefore capable of sustaining a nuclear chain reaction in the presence of energy neutrons. A process known as enrichment is used to isolate the ^{235}U isotope from uranium ore for use as fuel in nuclear reactors. The enrichment process produces enriched uranium (uranium hexafluoride with enhanced ^{235}U

concentration) and depleted uranium (uranium hexafluoride with reduced ^{235}U concentration). The enriched uranium is more radioactive and the depleted uranium is less radioactive than natural uranium (Weigel 1983).

The mass composition of depleted uranium (DU) is almost entirely ^{238}U (99.8%) with nearly all the ^{234}U (0.0006%) and approximately two thirds of the ^{235}U (0.2%) removed. DU radioactivity is approximately 60% that of natural uranium (Betti 2003). In terms of chemical, physical and toxicological behaviour, DU is the same as the metallic form of natural uranium (Harley *et al.* 1999; WHO 2001a,b).

Uranium is both a chemical and a radioactive material with variable chemical and physical forms. As such, uranium may be measured in units of mass or radioactivity (e.g., becquerel). A becquerel (Bq) is the amount of radioactive material in which one transformation (disintegration) occurs every second (ATSDR 1999). The current assessment deals only with the chemical aspects and hazards of naturally occurring uranium (expressed in units of mass), and generally excludes data on radioactivity. The naturally occurring uranium radionuclides have long half-lives and therefore, relatively low specific activities not usually associated with radiological health effects (Health Canada 1995).

Analytical Methods

A summary of the analytical methods for determining total uranium in environmental media (air, water, soil, sediment, biota, and vegetation) is provided in Table 4. Chemical methods utilized in uranium analysis include spectrophotometry, fluorometry, and kinetic phosphorescence. More recently, various mass spectrometer applications, including inductively coupled plasma-mass spectrometry (ICP-MS), atomic emission spectrometry (AES), mass spectrometry (MS) and accelerator-MS have been employed. Alpha (α) spectrometry is the analytical method commonly used for uranium isotope quantification (ATSDR 1999).

Scott (1973) lists sampling methods for uranium in occupational environments and Beverly and Ernstberger (1986) report methods for monitoring stack emissions.

Analysis of uranium in food has been conducted through preconcentration methods followed by alpha spectrometry or fluorometry (Sill 1977; Singh and Wrenn 1983). Dang and Chatt (1986) developed a simple and rapid method for the separation and determination of uranium from food by co-precipitation followed by neutron activation analysis (NAA).

Production and Uses in Canada

Uranium is mined as uranium ore, of which the predominant forms are carnotite, pitchblende, tobernite, uraninite, uranophane, davidite, and autonite. Commercially important mines are located in northern Saskatchewan, Canada; the Rand gold fields in South Africa; as well as sites in Colorado and Utah (United States), Australia and France (Merck Index 1989, G. Bird, pers. comm. 2001). Currently, open-pit mining, *in situ* leaching, and underground mining are three techniques for the mining of uranium-containing ores (ATSDR 1999).

Uranium ores in Canada are found mainly in northern Saskatchewan, Ontario, Quebec, New Brunswick, and the Northwest Territories. Minor deposits are located in British Columbia and Labrador (Energy, Mines and Resources Canada 1981). Northern Saskatchewan high-grade ores can contain an average of 15% uranium (G. Bird, pers. comm. 2001). The last operating uranium mine in Ontario was closed in 1996 (NRCan 2000).

Saskatchewan was the world leader in uranium production and accounted for 100% of Canadian production in 2001 and 2002. Total production in 2001 was 12 586 tonnes (t) of uranium from operations in the Athabasca basin of northern Saskatchewan (Cluff Lake, Mclean Lake, Key Lake/McArthur River, and Rabbit Lake). Although mining ceased at 2 operations in 2002 (Clugg Lake and Mclean Lake), stockpiled ore was milled until the end of 2002 and a new mine (Cigar Lake) is targeted for production in 2006. The majority of uranium reserves in the Athabasca basin are controlled by Cameco Corporation (57%) and COGEMA Resources Inc. (35%) (NRCan 2003).

Uranium occurs in phosphate rock from Florida, Texas, and southeastern Idaho at concentrations up to 120 ppm (NRCP 1975) and is released as a result of mining for phosphorous, which is then used in phosphate fertilizers (NCRP 1984). In the U.S., trace amounts of uranium progeny in phosphate fertilizer results in the distribution of about 120 Ci (180 metric tons) uranium over U.S. agricultural lands (Kathren 1984). Annual phosphate fertilizer consumption in Canada for 2001-2002 was reported to be 1,238,170 tonnes (Korol 2002). Canada has one phosphate mine in Kapuskasing, Ontario, which began operations in 1999. Phosphate rock was previously imported from Africa (Togo) for processing in Alberta (Korol and Larivière 1998). No estimates were found for uranium content in phosphate rock from either mine site.

The use of uranium in the generation of nuclear power results in production of depleted uranium (DU) hexafluoride, a by-product of the uranium enrichment process. Depleted uranium hexafluoride is converted to uranium metal (Weigel 1983). Due to its high density (twice that of lead) DU has been used in munitions and to reinforce military vehicles. Civilian uses include ballasts in aircraft, radiation shields in medical equipment, chemical catalysts, glassware, ceramics, and dentistry (Betti 2003).

DU is also used in armour-piercing ammunition, internal guidance devices and gyro compasses, counterweight devices for missile re-entry vehicles, radiation shielding material, and x-ray targets (Merck Index 1989). DU may be used to prepare alloys which are then further processed into steel (Environment Canada 1983). Very small amounts are used in light bulbs, photographic chemicals and household products (ATSDR 1999).

Sources and Concentrations in the Environment

The assessment of soil quality for naturally occurring metals must take into consideration regional variations in background concentrations in Canada. Background concentrations and environmental fate of metals strongly depends on geological and biological characteristics and therefore, any assessment of potential risks should take into consideration regional differences in

metal content in the natural environment (Chapman and Wang 2000).

Relatively high concentrations of metals can occur naturally in Canadian soils, stream sediments, and water, blurring the distinction between anthropogenic pollution versus naturally occurring bodies of ore. Uranium ore bodies identified in Canada include: Blizzard and Prairie Flats (British Columbia); Midwest Deposit (Athabasca region of northern Saskatchewan); Bancroft and South March (Ontario); Kaipokok (Labrador), and; Lineament Lake (Northwest Territories).

Soils and sediments reflect the composition of parent material, resulting in higher metal concentrations in mineralized areas (Wilson et al. 1998). Mining districts are characterized by naturally occurring metals in soil, sediment, rock, and water at concentrations that could result in their classification as "contaminated sites" (Painter et al. 1994). In the determination of anthropogenic metal contamination of soils, no single guideline concentration can adequately represent the variance in background concentrations across Canada (Painter et al. 1994; Chapman and Wang 2000).

A national ecological framework has been developed for Canada to encourage a more comprehensive approach to monitoring and reporting on the environment (Marshall and Schut 1999). Using various levels of ecoclassification, this framework identifies differences in ecological processes and character of a region (e.g., from the influence of agriculture on Prairie grasslands to the urban development of southern Ontario). Regional differences were considered for uranium content in stream geochemical surveys conducted across Canada, as well as in sediment or soil data for Quebec, Manitoba, Saskatchewan, and the Yukon.

Data representing background or ambient concentrations of total uranium in the Canadian environment are presented for air, surface water, sediments, drinking water, soil, dust, groundwater, biota, and biota used as human food. Where available, data on naturally occurring anomalous concentrations of uranium in soil, water, sediments, and biota were also provided for uranium ore bodies located in British Columbia, Saskatchewan, Ontario, Northwest Territories (NWT), and Labrador.

In some cases, the data reported came from published sources; however, information was also provided as unpublished (raw) data and may be subject to potential errors and omissions. The monitoring data provided here are dynamic and should be considered representative of the best available information for the time period reported. All references have been provided for unpublished data cited in this report.

Atmosphere

There were limited data available for ambient atmospheric uranium concentrations in Canada. Uranium is not monitored through the federal National Air Pollution Surveillance (NAPS) network of monitoring stations, nor is it a component of provincial monitoring in Alberta, Ontario, or Quebec (M. Peris, Environment Canada, pers. comm; B. Myrick, Alberta Environment, pers. comm.; OME 2004; M. Bisson, Quebec Ministry of Environment, pers. comm.).

Typical atmospheric uranium concentrations were reported to range from 0.025 to 0.1 ng/m^3 (NCRP 1999). A mean concentration of 0.076 ng/m^3 was reported for New York City, based on two composite samples from weekly air sampling conducted in 1985/1986 (Fisenne *et al.* 1987).

Stevenson and Pan (1996) reported that average total uranium activity levels within the continental United States dropped from nearly 2 μBq/m^3 (or 0.05 ng/m^3) in 1976 to below 1 μBq/m^3 (or 0.025 ng/m^3) in 1993.

Annual mean concentrations of uranium in air (probably present as particulates) near a Canadian uranium refinery in Port Hope, Ontario varied from 2 to 4 ng/m^3 (Cameco Corporation 2002). Ahier and Tracy (1997) reported that Port Hope air within 2 km of a uranium refinery varied from < 0.06 ng/m^3 to 76 ng/m^3 with a geometric mean of 1.0 ng/m^3. During summer slowdown periods, Ahier and Tracy (1997) reported a decrease in air concentrations of uranium which is suggestive that airborne uranium was primarily due to processing operations rather than resuspension of soils containing previously emitted uranium. The background concentration of uranium in southern Ontario has been reported to be about 0.1 ng/m^3 (Tracy and Prantl 1985).

Soil and Dust

Uranium occurs as a natural constituent in soil, originating from rocks in the earth's mantle. The basic rocks (basalts) contain <1 mg/kg uranium; while acidic rocks (granites) contain >8 mg/kg; and sedimentary rocks (shale) contain, on average, 4 mg/kg of uranium (Stokinger 1981; NCRP 1984). The abundance of uranium in the earth's crust ranges from 0.00027 to 0.0004% (Riley and Chester 1971; Berlin and Rudell 1979), with an average concentration of 3 to 4 mg/kg (Statistics Canada 1983).

Worldwide uranium concentrations in soil have been reported to range from 0.3 to 11.7 mg/kg (UNSCEAR 1993). The average background concentration of uranium in soil is about 2 mg/kg (NCRP 1984). Data specific to uranium concentrations in Canadian soils were identified for B.C., Saskatchewan, Manitoba, Ontario, and New Brunswick (Table 5).

In a comprehensive literature review of uranium exploration studies, Gordon (1992) reported the link between ore bodies and biosphere concentrations of uranium at six locations in Canada: Blizzard Deposit, B.C.; Midwest Deposit, Saskatchewan; Bancroft Deposit, Ontario (mining area); South March Deposit, Ontario; Lineament Lake Deposit, NWT, and; Kaipokok Deposit, Labrador.

The Blizzard uranium deposit occurs in the Okanagan Highlands, in the eastern Tectonic Belt of the Western Cordillera, approximately 50 kilometers southeast of Kelowna, B.C. Uranium in soil samples collected on or within 13 km of the ore body in spring/summer (n=64) ranged from 1.5 to 390 mg/kg, with a typical range of 5 to 15 mg/kg reported. In upland till, lowland till and lowland alluvium samples on or within 13 km of the ore body (n=11), a range of <0.5 to 125 mg/kg and an overall mean of 25 mg/kg was reported; the highest uranium concentrations occurred in lowland alluvium. Generally, uranium concentrations in soils collected off the ore

body ranged from 3 to 7 mg/kg (Gordon 1992).

The Midwest uranium deposit in Northern Saskatchewan occurs in the Athabasca basin. In basal till samples collected near and under the southern shore of S. McMahon Lake, including mineralized boulder train, uranium concentrations ranged from non-detectable to 38 mg/kg (n=361). In soil horizon Ah (0-40 cm), uranium concentrations ranged from 0.4 to 6.3 mg/kg, with an overall mean of 1.6 mg/kg (Gordon 1992).

In Port Hope, Ontario, a total of 74 surface and sub-surface samples from 10 sites subjected to potential airborne uranium deposition were obtained (Sheppard *et al.* 2004). The samples were digested in aqua- regia and analysed by ICP-MS. The uranium levels ranged from 0.6 to 258 mg/kg with a mean and median of 13.9 mg/kg and 1.7 mg/kg, respectively.

The Bancroft uranium deposit (Ontario) occurs within the Grenville Province of the Canadian Shield, it is an active mining area and well known for natural occurrences of uranium. Soil concentrations ranged from <1 to 500 mg/kg in A horizon soils (n=25) and from <1 to 70 mg/kg in B horizon soils (n=21) over the Faraday ore body (Gordon 1992).

The South March uranium site is located in the Ottawa-St. Lawrence lowlands, approximately 20 km west of Ottawa, Ontario. A maximum uranium concentration of 110 mg/kg was reported within a 1.7 km^2 grid area (n=371) and a baseline study reported a maximum uranium concentration of 60 mg/kg for soil samples (n=34) collected along 2 mineralized sandy dolomite horizons. The background concentration for the region was reported to be 1.9 mg/kg. No soil concentration data were reported for the Lineament Lake uranium ore deposit of the NWT or the Kaipokok uranium deposit of Labrador (Gordon 1992).

Total uranium concentrations determined by fluorimetry in soil samples collected at the Prairie Flats surficial uranium deposit near Summerland B.C. ranged from 3 to 572 mg/kg (Van Netten and Morley 1982b). The soil samples were chosen to represent the range of uranium concentrations found in the deposit. A value of 3 mg/kg might be considered representative of background uranium concentrations for the area.

Uranium concentrations were measured in a bog habitat and a Jackpine habitat within a control (background) site near the Key Lake uranium mill in northern Saskatchewan (Thomas 2000). Soil samples were analysed for uranium by delayed neutron counting in a Slowpoke II reactor. Three samples were generally collected for each soil layer and all results were in dry weight. The mean uranium concentration in the upper peat layer (0-10 cm) of the bog habitat was 3.5 mg/kg. In the upper surface soils (0-2 cm) of the Jackpine habitat, a mean uranium concentration of 1.1 mg U/kg (n=3) was determined, while slightly lower down (2-10 cm) a mean of 0.53 mg/kg (n=3) was reported. Uranium concentrations in soil were elevated compared to previously reported baseline ranges for the area (Key Lake Mining Corp 1979; Takala 1991; Takala 1994; Thomas 1997).

In rural northern Manitoba, uranium concentrations (dry weight) in background surface soils (n=6) ranged from 0.43 to 1.29 mg/kg (E. Yee, Manitoba Conservation, pers. com). An overall

average of 0.65 mg U/kg was calculated.

A multi-element profile of indoor dust in relation to outdoor dust and garden soils (0-5 cm) was conducted for the city of Ottawa, Ontario (Rasmussen *et al.* 2001). This city represents an urban centre with a low concentration of heavy industries. Random samples of house dust as well as street dust and garden soil (within 15 m of each residence) were collected from 10 zones in the city of Ottawa. Metal content (dry weight) was determined by inductively coupled plasma mass spectrometry (ICP-MS). The minimum detection limit for uranium was 0.01 mg/kg. Uranium concentrations in soil ranged from 0.66 to 2.64 mg/kg (n=50). The concentration representing the 95th percentile was 1.96 mg/kg. An arithmetic mean of 1.17 mg/kg and a geometric mean of 1.11 mg/kg were reported. The uranium concentrations in adjacent street dust samples (n=45) ranged from 0.43 to 2.25 mg/kg, with a 95th percentile concentration of 1.02 mg/kg and arithmetic and geometric mean values of 0.82 and 0.79 mg/kg, respectively. In house dust (n=48), uranium concentrations ranged from 0.29 to 1.33 mg/kg with a 95th percentile concentration of 1.06 mg/kg and arithmetic and geometric mean values of 0.58 and 0.55 mg/kg, respectively.

In 1990, surface soils (0-5 cm) were sampled from 12 urban locations in Windsor and from 18 rural locations in Essex County (Gizyn 1994). The soil samples were collected as part of a baseline study of soil, produce and air quality prior to operation of the Detroit municipal waste incinerator. In urban soils, uranium concentrations (dry weight) ranged from 0.4 to 1.3 mg/kg with an arithmetic mean of 0.9 mg/kg. Uranium concentrations in rural soils were slightly higher, ranging from 0.8 to 2.2 mg/kg with an arithmetic mean of 1.2 mg/kg.

In Ontario, surface soils (0-5 cm) not impacted by point sources of pollution from old urban parkland sites and rural parkland sites were analysed for uranium using ICP-MS following nitric acid (HNO_3) digestion (OMEE 1993). In old urban parkland soils (n=60), uranium concentrations (dry weight) ranged from 1.7 (lower concentration limit or LCL) to 2.2 mg/kg (upper concentration limit or UCL). Rural parkland concentrations (n=17) ranged from 0.8 (LCL) to 1.5 mg/kg (UCL) soils in the NW region. For rural parkland soils in the SW, west central, central, SE, and NE regions of the province (n=84), concentration ranges from 2.0 (LCL) to 9.7 mg/kg (UCL) were reported. The Ontario Typical Range (OTR) 98th percentile values, encompassing 98% of the uranium concentrations in the sites sampled, were 1.9 mg/kg for old urban parkland soils and 2.1 mg/kg for rural parkland soils (OMEE 1993).

A regional background value of 33 mg U/kg was recommended for soils in the Port Hope urban area. This concentration represents the 97.5th percentile value of uranium surface soil concentrations in areas unaffected by direct atmospheric deposition of low level radioactive waste from the Cameco uranium mine. However, historical atmospheric fallout from previous mining activities (Eldorado's Gold Mine radium refinery) may have contributed to these surface soils (Stantec 2004).

Metal concentrations in garden soils were measured as part of a multi-media analysis of heavy metals in urban and rural sites in New Brunswick (Pilgrim and Schroeder 1997). Uranium concentrations (dry weight) were determined by ICP-MS in surface soil samples collected from 9 urban gardens in East Saint John (ESJ), 2 urban gardens in West Saint John (WSJ), and 1 rural

garden in Fredericton. Mean uranium concentrations for soils from ESJ, WSJ, and rural Fredericton were 1.9 mg/kg (n=18), 1.8 mg/kg (n=4) and 2.3 mg/kg (n=2), respectively.
In an analysis of Port Hope soils, a total of 74 surface and sub-surface samples from 10 sites subjected to potential airborne U deposition were obtained (Sheppard *et al.*, 2004). The samples were digested in aqua- regia and analysed by ICP-MS. The U levels ranged from 0.6 to 258.3 mg/kg with a mean and median of 13.9 mg/kg and 1.7 mg/kg, respectively.

Groundwater

Naturally occurring uranium concentrations in groundwater are generally low, typically less than 1 µg/L but may vary considerably with much higher levels in private and community wells across Canada (Health Canada 1995; 1999).

Uranium concentrations were provided for groundwater samples collected in various locations on B.C.'s gulf islands and Vancouver Island (Nanaimo, Gabriola Island, Qualicum, and Galiano Island,) (P. Evans, BCMWLAP, pers. comm.). Total uranium concentrations (n=5) ranged from 0.30 to 0.50 µg/L, with an overall average of 0.38 µg/L.

In southeastern Manitoba, samples (n=287) collected from 1982 to 1984 contained mean and median uranium concentration of 58.3 µg/L and 10 µg/L, respectively (Betcher *et al.* 1988).

Groundwater was evaluated for uranium isotopes in a region of Nova Scotia with naturally elevated uranium in the bedrock (Avon Valley) (Kronfeld 2004). Samples (n=20) collected from domestic wells in 1998/1999 were analysed for ^{238}U concentrations using alpha spectrometry. Concentrations were typically <1.0 µg/L (85%) and ranged from 0.024 to 41 µg ^{238}U/L.

The release of uranium from anthropogenic sources in groundwater results primarily from ore production and from the disposal of solid wastes from mining, milling and production operations. Phosphate fertilizers may contain uranium and can also contribute uranium to groundwater (Spalding and Sackett 1972).

Groundwater collected within an 18 km^2 radius of the Blizzard (B.C.) uranium deposit contained from 0.65 to 85 µg U/L, with a mean value of 18.79 µg U/L (n=25) reported (Gordon 1992).

Groundwater was collected at the Prairie Flats surficial uranium deposit near Summerland B.C. in September 1997 and March 1998 and analysed for uranium using ICP-MS. Uranium concentrations ranged from 9 to 3961 µg/L in groundwater from 9 shallow wells (peat and clay unit at 1.5 m) and from 12 to 743 µg/L in groundwater from 4 deep wells (sand and gravel unit at 3 m). Measured vertical hydraulic gradients indicated a significant upward discharge of groundwater into the peat and clay unit (Tixier and Beckie 2001).

Uranium concentrations in groundwater (n=130) collected in a regional study area (22,000 km^2) of the South March uranium site (Ontario) ranged from <0.2 to 73.0 µg/L, with a mean value of 1.4 µg/L reported (Gordon 1992).

Groundwater data were provided for total uranium in raw water samples (n=910) collected across Ontario (P. Cheung, Ontario MOE, pers. comm.). Data from 1998 to 2002 were available for groundwater supply wells under the Ontario Drinking Water Surveillance Program. Annual average uranium concentrations were reported for each well (n=51) and ranged from 0.005 to 11.5 µg/L, with an overall mean of 0.8 µg/L.

Surface Water

Environmental concentration ranges previously reported for uranium in Canadian surface waters were 0.10-2.1 µg/L for the Pacific region; 0.097-2.14 µg/L for the western region; 0.28-0.65 µg/L for the central region, and; 0.25-0.73 µg/L for the Atlantic region (CCREM 1987). More recent monitoring data for uranium concentrations in Canadian surface waters are provided below. In many cases, it is difficult to determine whether concentrations measured represent natural background levels, or whether there have been inputs from anthropogenic sources.

Surface waters (lakes and streams) collected within a 10 km radius of the Blizzard (B.C.) uranium deposit (n=115) contained from 0.05 to 1.70 µg U/L, with a mean value of 0.44 µg U/L (Gordon 1992).

Surface water data were provided for 5 river basins in Alberta (R. Tchir, Alberta Environment, pers. comm.). Total uranium concentrations from 1999 to 2002 were available for all the rivers analysed. Uranium samples were digested with nitric acid and total uranium was determined using ICP-MS. In the Athabasca river basin, uranium concentrations ranged from <0.4 to 1.0 µg/L, with an average concentration of 0.6 µg/L (n=49). In the North Saskatchewan-Battle river basin (n=91), an average concentration of 0.7 µg/L (<0.4 to 6.2 µg/L) was reported. Uranium concentrations in the Bow river basin ranged from <0.4 to 5.1 µg/L, with an average value of 0.8 µg/L (n=198). The average concentration for the Red Deer river basin (n=45) was 0.9 µg U/L (ranging from <0.4 to 2.3 µg U/L) and the average concentration in the Old Man River basin (n=108) was 1.6 µg U/L (<0.4 to 9.8 µg/L).

Data on uranium concentrations in Saskatchewan surface waters were provided for lakes, rivers and streams (S. Hase, Saskatchewan Environment, pers. comm.). Samples were analysed by direct fluorometry or by solvent extraction followed by fluorometry. Annual average uranium concentrations reported for 1998 to 2000 ranged from 0.5 µg/L to 19.4 µg/L in rivers and streams (n=32). In lakes, annual average concentrations reported for 1998 to 2001 ranged from 0.6 µg/L to 3.5 µg/L (n=45).

Lake waters sampled within a 24 km^2 grid of the Midwest uranium deposit (Northern Saskatchewan) (n=36) contained between 0.05 to 0.48 µg U/L, with an overall mean of 0.07 µg U/L. Sampling of anomalous areas in the same region (n=44) yielded a uranium concentration range from 0.05 to 3.0 µg/L, with an overall mean of 0.18 µg/L. A uranium concentration of 0.1 µg/L was estimated for background surface water in this region (Gordon 1992).

Uranium concentrations measured in background surface waters (lakes and rivers) in rural Northern Manitoba (n=5) were below analytical detection or <0.1 µg/L (E. Yee, Manitoba Conservation, pers. com).

Regional surface water samples from creeks, rivers, lakes and swamps were collected over a 4500 km^2 region around the Bancroft (Ontario) uranium deposit (n=1,110). Uranium concentrations ranged from <1 to 700 µg/L, with an overall average of 1.6 µg/L. A background uranium concentration range of 1 – 2 µg/L was reported for surface water in this region (Gordon 1992). In surface water samples collected in 1990 and 1991 from control lakes (not impacted by mining activity) near the city of Elliot Lake, Ontario, mean total uranium concentrations (ICP-MS) ranged from <1 to 1 µg/L (Clulow *et al.* 1998).

Data on total uranium concentrations in surface waters (1992 and 1993) were provided for Port Hope Harbour, Lake Ontario (D. Boyd, Ontario Ministry of the Environment, pers. comm.). Total uranium concentrations for samples collected in proximity to the harbour entrance (100 metres south, 50 m east, and 50 m west) ranged from <0.1 to 1.7 µg/L (n=72).

Uranium concentrations were measured in 1,522 samples of raw water from lakes (n=841) and rivers (n=681) collected in Ontario between 1998 and 2002. Samples were collected from 78 lake water treatment plants and 45 river water treatment plants, under the Ontario Drinking Water Surveillance Program (P. Cheung, Ontario MOE, pers, com.). Average uranium concentrations were reported over this time period for each treatment plant. In raw lake water, average uranium concentrations ranged from 0.007 to 3.2 µg/L. In river water, average concentrations ranged from 0.03 to 1.8 µg/L.

Surface waters were evaluated for uranium isotopes in a region of Nova Scotia with naturally elevated uranium in the bedrock (Avon valley) (Kronfeld 2004). Samples of lake and stream waters were collected in 1998/1999 from the Avon River drainage basin and were analysed for ^{238}U concentrations using alpha spectrometry. A ^{238}U concentration of 0.29 µg/L was determined for lake water (n=1) and concentrations ranging from 0.15 to 0.28 µg ^{238}U/L were determined for streams (n=4).

Surface water concentrations in the Kitts study area (32 km^2) of the Kaipokok uranium deposit (Labrador) ranged from 0.1 to 255 µg U/L (n=9) (Gordon 1992).

In a review of arctic marine ecosystem contamination, Muir *et al.* (1992) reported total uranium concentrations in surface waters of northern lakes. In the NWT, a total uranium concentration of 0.16 µg/L was reported for Baker Lake (Baweja *et al.* 1987), while uranium concentrations in Sherman Lake (mining site) and Great Slave Lake (Port Radium Area) ranged from 1 to 5 µg/L, and from 1 to 400 µg/L, respectively (Veska and Eaton 1991; Kalin 1984). Uranium concentrations in Beaverlodge Lake, northern Saskatchewan (mining site) were reported to range between 200 to 400 µg/L (Swanson 1983). A Canada-wide background range of 0.2 to 5 µg/L was reported for total uranium in surface waters (Baweja *et al.* 1987).

A median river water concentration of 0.05 µg U/L was reported for the Yukon (Northern Shelf geological region), based on samples (n=875) collected from stream geochemical surveys (Heon 2003).

Uranium concentrations in surface water from the Lineament Lake uranium deposit of the NWT ranged from non-detectable to 1.0 µg/L, with an overall mean of 0.2 µg/L (n=257). A regional background concentration of 0.2 µg U/L was reported (Gordon 1992).

Uranium additions to the world's oceans by river flows and sediments are about 11 000 and 8000 t annually, respectively (Goldberg 1976). No data, however, were found on uranium concentrations in Canada's coastal waters.
Anthropogenic sources of uranium in surface water are primarily due to ore production and disposal of solid wastes from mining, milling and production operations. Phosphate fertilizers may contain uranium and can also contribute uranium to groundwater (Spalding and Sackett 1972).

Environment Canada/Health Canada (2003) summarized reports that provided concentrations of uranium in surface water at background and near nuclear facilities in Canada. In northern Saskatchewan, background median surface water concentrations of uranium were less than the detection limit of 0.05 µg/L with a 95th percentile concentration of 0.35 µg/L. In regions surrounding Elliot Lake, Ontario, background median surface water concentrations of uranium were less than the detection limit of 0.05 µg/L with a 95th percentile concentration of 0.28 µg/L. Near uranium mines in Saskatchewan, a geometric mean uranium surface water value of 0.03 µg/L with a 90th percentile of 0.85 µg/L was reported with a maximum value of 0.85 µg/L. At decommissioned mines in Serpent River watershed/Elliot Lake area of Ontario, surface water concentrations ranging from 0.5 to 15.3 µg/L were reported. Surface water at decommissioned mines in the Beaverlodge Lake, Saskatchewan area had surface concentrations in the range of 59 to 168 µg/L.

Sediments

Mean concentrations of uranium in the Great Lakes sediments were previously reported to be 2.19 mg/kg in Lake Michigan, 0.59 mg/kg in Lake Superior, 0.43 mg/kg in Lake Erie, 0.41 mg/kg in Lake Huron, 0.38 mg/kg in Georgian Bay, 1.22 mg/kg in the St. Clair River; and 0.23 mg/kg in Lake Ontario (CCREM 1987). Ranges of concentrations of uranium in sediments previously reported for various regions of Canada were 22-24.7 mg/kg in Northwest Territories lakes; 8.99-13.7 mg/kg in British Columbia streams; 3.04-12 mg/kg in Yukon streams; 3.34-6.15 and 5.1 mg/kg in Saskatchewan lakes and streams, respectively; 5.08-5.35 mg/kg in Ontario lakes; and 1.47-2.38 mg/kg in Newfoundland lakes (CCREM 1987). More recent monitoring data are provided below for uranium concentrations in Canadian sediments.

A median value of 2.8 mg/kg and a 95th percentile value of 21.2 mg/kg were reported for uranium in lake and stream sediment samples (n= 222 and 192, respectively) collected across Canada (under the National Geochemical Reconnaissance program) and within Quebec (under the Quebec survey program) (Painter *et al.* 1994). According to the data collected under this

program, uranium concentrations above 20 mg/kg occurred in sediments sampled in Nunavut, the NWT, Yukon, B.C., Northern Manitoba, Northern Saskatchewan, Southern Ontario, Northern Quebec, and Newfoundland and Labrador (Figure 1).

Total uranium concentrations were measured (October 2001) in lake and river sediments in the Kootenay region (between West Arm of Kootenay Lake and the confluence with the Columbia River) of B.C. (BCMWLAP 2004). All concentrations were determined by ICP-MS and reported as dry weight. In the Columbia river, uranium concentrations of 2.5 mg/kg (Birchbank) and 2.7 mg/kg (Waneta) were reported. Uranium concentrations of 2.1 mg/kg (Glade) and 7.5 mg/kg (Taghum) were reported in sediments from the Kootenay River. Sediments from Arrow Lake contained 4.6 mg U/kg.

Lake and stream sediments collected within a 10 km radius of the Blizzard (B.C.) uranium ore deposit (n=10) contained uranium concentrations ranging from 2.1 to 21.6 mg/kg, with a mean value of 10.3 mg/kg (Gordon 1992).

Uranium concentrations in background surface sediment samples from lakes and rivers in Northern Manitoba (n=5) ranged from 0.34 to 0.85 mg/kg (E. Yee, Manitoba Conservation, pers. com).

Data on uranium concentrations in sediments collected in 1986 from St. Mary's river (Ontario) were provided by the Ontario Ministry of the Environment (D. Boyd, pers. comm.). Total uranium concentrations (dry weight) ranged from 0.4 to 1.1 mg/kg, with an average of 0.5 mg/kg.

Sediment samples collected in 1990 and 1991 from control lakes (not impacted by mining activity) near the city of Elliot Lake (Ontario) were analysed for total uranium (ICP-MS) (Clulow *et al.* 1998). Mean uranium concentrations of 4.1 mg/kg (Jimchrist Lake) and 54.1 mg/kg (Semiwite Lake) were reported for lake sediments.

Lake sediments sampled within a 24 km^2 grid of the Midwest uranium deposit area of the Athabasca basin (Northern Saskatchewan) contained between 0.9 to 13.2 mg U/kg (n=48). Sampling of anomalous areas (South McMahon Lake) in the same region yielded a uranium concentration range from 1.8 to 12.0 mg/kg, with an overall mean of 6.7 mg/kg. (n=66). A regional background concentration range of 1 to 2 mg U/kg was estimated (Gordon 1992).

Uranium concentrations in stream sediments collected in the uranium ore area of South March (Ontario) ranged from below analytical detection to 11.0 mg/kg (n=50). In Lineament Lake sediments (NWT) uranium concentrations ranged from below analytical detection to 70 mg/kg (n=1,349). A geometric mean of 0.7 mg U/kg and an arithmetic mean of 1.3 mg U/kg were reported. Lake sediments collected in the Kaipokok uranium deposit of Labrador contained uranium at concentrations from below analytical detection to 104 mg/kg (n=184) (Gordon 1992).

Uranium concentrations in stream and lake sediment samples (with some soil samples mixed in) were measured over a 30 year period (1967 to 1997) in five geological regions of Quebec,

including the Appalachians, St. Lawrence lowlands, Grenville, Upper and Rae, and the Labrador Trench (Choinière and Beaumier 1997). The geometric means reported for uranium ranged from 0.66 to 2.93 mg/kg (n= 39,016) in the Appalachian region, from 0.37 to 0.74 mg/kg (n=910) in the St. Lawrence lowlands, from 1.01 to 2.89 mg/kg (n=37,681) in Grenville, and from 0.6 to 3.33 mg/kg (n=22,726) in Upper and Rae. A geometric mean of 1.82 mg/kg was reported for uranium in the Labrador Trench. Limitations to this data include the variation in materials analysed (sediment and soil) and differences in the analytical methods used over time. Samples collected up until the early 1980's were analysed by atomic absorption. This technique was gradually replaced with plasma-atomic emission spectrometry (ICP-AES), resulting in a more sensitive detection limit and a higher degree of confidence in the lower percentiles of reported distributions (Choinière and Beaumier 1997).

Data collected from stream geochemical surveys conducted in the Yukon were reported for 7 "geological provinces" within the territory (Heon 2003). Sediments were analysed by neutron activation followed by delayed neutron counting (NADNC) or ICP-MS. Yukon-wide, uranium sediment concentrations ranged from 0.1 to 481 mg/kg, with a median value of 3.3 mg/kg (n= 27,161 uranium samples). Median uranium concentrations reported for each geological province were as follows: Cassiar Platform (4.9 mg/kg, n=2318), Insular (1.7 mg/kg, n=398), Intermontane (2.5 mg/kg, n=2325), North-American Shelf (2.7 mg/kg, n=4765), Selwyn Basin (4.4 mg/kg, n=5953), Triass-Cretac (n=2.3 mg/kg, n=1840), and Tanana Terrane (3.3 mg/kg, n=7499).

Environment Canada/Health Canada (2003) summarized reports that provided concentrations of uranium in sediments at background and near nuclear facilities in Canada. In northern Saskatchewan, a background median sediment concentration of uranium equal to 3.7 mg/kg and a 95th percentile concentration of 29.5 mg/kg were reported. While, in regions surrounding Elliot Lake, Ontario, a median concentration of 4.2 mg/kg and a 95th percentile concentration of 51 mg/kg were reported for uranium in sediment. Near uranium mines, a geometric mean uranium sediment value of 2.63 mg/kg with a 90th percentile of 9.15 mg/kg was reported with a maximum value of 648 mg/kg. It was concluded by Environment Canada/Health Canada (2003) that baseline sediment concentrations of uranium near uranium mines in northern Saskatchewan are relatively low. At decommissioned mines in Serpent River watershed/Elliot Lake area of Ontario, sediment concentrations ranging from 3.4 to 270 mg/kg were reported. Sediment concentrations at decommissioned mines in the Beaverlodge Lake, Saskatchewan area had sediment concentrations in the range of 35 to 2,260 mg/kg.

Drinking Water

Between 1975 and 1985, uranium concentrations in drinking water supplies of up to 17 Canadian cities were monitored. Reported concentrations for the monthly composite samples are usually less than 1 µg/L, but there have been values greater than 8 µg/L (DNHW 1985). In 1980, the average uranium concentration for groundwater in Canada was reported to be 0.2 µg/L (Health and Welfare Canada 1980). About 50% of the uranium concentrations in monthly samples from 17 municipalities were below the detection limit of 0.05 µg/L (DNHW 1985). Higher concentrations exist in the Tweed area of southeastern Ontario (up to 80 µg/L) (Environment

Canada 1983), and in southwestern Saskatchewan (up to 39 µg/L) (Dyck *et al.* 1976). In another study, private domestic supplies with uranium concentrations were up to 700 µg/L in Canada (Moss *et al.* 1983; Moss 1985). Recent monitoring data for uranium concentrations in Canadian drinking water are provided below

Drinking water data were provided by the Yukon Department of Environmental Health (E. Bergsam, pers. comm.). Total uranium concentrations reported in samples collected in 2002 (n=18) ranged from <0.5 to 7.2 µg/L.

Drinking water from groundwater wells and treatment plants were sampled in 2002 in rural Northern Manitoba (E. Yee, Manitoba Conservation, pers. com). Uranium concentrations in well water (n=8) ranged from <0.1 to 4.6 µg/L. In water collected from treatment plants (n=3), uranium concentrations ranged from <0.1 to 0.2 µg/L.
Uranium concentrations in drinking water samples from rural communities in Manitoba (n=68) ranged from below detection (<0.0005 µg/L) to 0.013 µg/L (Yuen and Zimmer 2001). In the majority of samples (~70%) uranium concentrations were below analytical detection.

In Nova Scotia, uranium concentrations in public well water were tested in 52 schools in 2002 (NSDEL 2002). Two of the water well samples were found to exceed the Guideline for Canadian Drinking Water Quality of 20 µg/L (i.e., 27 and 81 µg/L).

Drinking water data for total uranium were provided for 356 samples of distribution and treated water from groundwater (n=98), lake (n=160), and river (n=98) sources in Ontario (P. Cheung, Ontario MOE, pers. comm.). Data for 1998 to 2002 were provided by 80 lake water treatment plants, 49 river water treatment plants, and 53 groundwater supply wells under the Ontario Drinking Water Surveillance Program. Average concentrations were reported by each station over this time period; overall averages calculated for uranium were 0.8 µg/L (average concentrations ranged from <0.002 to 11.7 µg/L) in groundwater, 0.2 µg/L (<0.002 to 4.1 µg/L) in lake water, and 0.2 µg/L (0.005 to 1.1 µg/L) in river water.

Uranium concentrations in potable domestic well water samples collected from 1974-1982 were reported for four geological regions of Quebec (Choinière and Beaumier 1997). Extensive sampling resulted in the following geometric means reported for potable groundwater: Appalachians (0.44 µg/L, n=14,756), St. Lawrence lowlands (0.97 µg/L, n=6,716), Grenville (0.35µg/L, n=1,709), and Upper and Rae (0.65 µg/L, n=5,482). Samples were primarily analysed by atomic absorption, resulting in a less sensitive detection limit which affects the degree of confidence in the lower percentiles of the reported distributions (Choinière and Beaumier 1997).

Biota

Uranium concentrations in aquatic vegetation (n=9) within 13 km of the Blizzard uranium ore deposit (Okanogan, B.C.) ranged from 0.075 to 2.10 mg/kg with a mean value of 0.44 mg/kg reported (Gordon 1992).

Vegetation was sampled in the Midwest uranium deposit area of the Athabasca basin (Northern Saskatchewan). In spruce trunks (300 sites) uranium concentrations were below analytical detection (<0.2 mg/kg). Uranium concentrations ranged from 8 to 46 mg/kg) in Labrador tea stems and from 4 to 39 mg/kg in Labrador tea leaves. Spruce twigs (69 sites) contained the highest uranium concentrations, ranging between 20 to 130 mg/kg (Gordon 1992).

Uranium concentrations were measured in vegetation, small mammals and birds from a bog habitat and a Jackpine (upland) habitat located within a control (background) site near the Key Lake uranium mill in northern Saskatchewan (Thomas 2000). For vegetation, three samples were generally collected with all of the results reported as dry weight. Mean uranium concentrations in the bog habitat were reported for young black spruce needles (0.069 mg/kg), old black spruce needles (0.370 mg/kg), young black spruce twigs (0.760 mg/kg), old black spruce twigs (2.2 mg/kg), leatherleaf and Labrador tea (0.770 mg/kg), and bog litter (2.9 mg/kg). In the Jackpine habitat, mean uranium concentrations were reported for Jackpine needles (0.082 mg/kg), Jackpine twigs (0.220 mg/kg), blueberries (1.1 mg/kg), and upland litter (3.1 mg/kg). These vegetation results were within previously reported baseline ranges for the area (Key Lake Mining Corp 1979; Takala 1991; Takala 1994; Thomas 1997). Data were available for uranium concentrations (whole body, wet weight) in small mammals in northern Saskatchewan, although the information is limited by small sample size (Thomas 2000). The highest uranium concentration of 96 ppb (0.096 mg/kg) occurred in deer mice (n=1). A whole body uranium concentration of 35 ppb (0.035 mg/kg) was reported for Redbacked voles (n=5), and uranium concentrations of 21 ppb (0.021 mg/kg), 9 ppb (0.009 mg/kg), and 8 ppb (0.008 mg/kg) were reported for heather voles (n=1), meadow voles (n=2) and masked shrews (n=1), respectively. Birds inadvertently trapped in mouse traps were also analysed for uranium, however, concentrations were below analytical detection (<100 ppb or 0.1 mg/kg) in Lincoln's sparrow (n=1) and swamp sparrow (n=1). An overall mean uranium concentration of 24.5 ppb (0.0245 mg/kg and a range of 6 to 34 ppb (0.006 to 0.034 mg/kg) were reported for all small mammals. By comparison, a lower mean uranium concentration of 7.5 ppb (0.0075 mg/kg) and range of <0.5 to 20 ppb (<0.0005 to 0.020 mg/kg) were reported for small mammals collected from the southern prairies of Saskatchewan, away from uranium ore bodies and processing facilities (Thomas 1995).

Lichen (*Cladina stellaris*) collected around Wollaston Lake (Northern Saskatchewan) contained uranium (total) at concentrations ranging from 77 to 1,400 ppb (0.077 to 1.4 mg/kg, n=7), with a mean value of 229 ppb (0.229 mg/kg) (Thomas and Gates 1999).

Metal content was determined in vegetation collected from rural gardens in Northern Manitoba representing background locations. Uranium concentrations (dry weight) were reported for radishes (<0.009 mg/kg), potatoes (<0.006 mg/kg), carrots (<0.006 mg/kg), turnips (0.047 mg/kg), strawberries (<0.006 mg/kg), blueberries (<0.006 mg/kg), and mossberries (<0.006 mg/kg) (E. Yee, Manitoba Conservation, pers. comm.).

Vegetation biomonitoring has been conducted to evaluate metal content in air and airborne deposition. Tree foliage sampling data (dry weight basis) were available for background locations in rural northern Manitoba. Uranium concentrations in coniferous trees (n=3) sampled

in 2002 ranged from <0.006 to 0.012 mg/kg (dry weight) (E. Yee, Manitoba Conservation, pers. comm.).

Uranium concentrations in plant samples collected in the uranium ore area of South March (Ontario) ranged from 0.01 to 0.18 mg/kg (n=112) (Gordon 1992). Plant species were not reported, however, mixed forest vegetation (evergreens and deciduous species) occur in the region.

Uranium was found to reach concentrations of 10-15% of dry cell weight in the yeast *Saccharomyces cerevisiae*, with optimal sorption occurring at pH 5.9-6.8 (Strandberg *et al.* 1981). Zooplankton from Lake Superior, Lake Michigan and Lake Erie had uranium concentrations of approximately 18 µg/kg (CCREM 1987).

An average uranium concentration of 3 µg/kg (dry weight) was reported for small whole forage fish (*Alosa pseudoharengus*, *Notropis hudsonius*, *Percopsis omiscomaycus*) in Lake Superior, Lake Michigan, and Lake Erie (CCREM 1987).

Uranium concentrations (measured as ^{238}U) up to 240 mg/kg were reported in vegetation growing near (within 600 m of) naturally occurring uranium outcrops near Baker lake (NWT). Plants taken from non-mineralized sites contained <0.5 mg/kg, suggesting that the vegetation near the outcrops were accumulating uranium (Thomas *et al.* 1992).

Biota Used as Human Food

A study of 18 caribou collected north of the community of Wollaston Post (Northern Saskatchewan) reported uranium concentrations in caribou tissues (Thomas and Gates 1999). Uranium was detected in liver tissues but was below analytical detection in the majority (>50%) of bone, kidney, and muscle tissue samples. A value equal to one-half the detection limit was used to represent non-detect results in order to calculate mean concentrations. Mean total uranium concentrations were reported for liver (2.3 ppb or 0.0023 mg/kg), bone (26 ppb or 0.026 mg/kg), kidney (18 ppb or 0.018 mg/kg), and muscle (1.6 ppb or 0.0016 mg/kg). Detection limits ranged from 0.1 to 100 ppb (0.0001 to 0.1 mg/kg) depending on the tissue analysed. Total uranium was comprised of 0.711% ^{235}U, 99.2837% ^{238}U, and 0.0053% ^{234}U by weight.

In a review of arctic marine ecosystem contamination, Muir *et al.* (1992) reported total uranium concentrations in fish from Northern lakes (NWT, Saskatchewan, and Ontario).
Mean total uranium concentrations of 0.08 and 0.06 mg/kg (wet weight) were reported for whitefish and white sucker, respectively, caught from Beaverlodge Lake (NWT) (Swanson 1983). In Northern Saskatchewan, a mean total uranium concentration of 0.01 mg/kg (wet weight) was reported for whitefish caught in Fredette Lake, similar to the mean values of 0.006 and 0.01 mg U/kg (wet weight) reported for whitefish and white sucker from Milliken Lake (Ontario) (Swanson 1983). Uranium concentrations in Northern pike and whitefish from Lake Athabasca (Northern Saskatchewan) were reported to be <0.0006 and 0.00005 mg/kg (wet weight), respectively (Waite *et al.* 1988).

Total uranium concentrations (ICP-MS) were reported in fish collected in 1990 and 1991 from control lakes not impacted by mining activity (JimChrist and Semiwite) near the city of Elliot Lake, Ontario (Clulow *et al.* 1998). Mean uranium concentrations reported in bone tissues of laketrout and whitefish were <0.05 and 0.08 mg/kg (dry weight), respectively. In gut contents, mean uranium concentrations of 0.07 and 0.18 mg/kg (dry weight) were reported for laketrout and whitefish, respectively. Uranium was not detected in the muscle tissues (<0.05 mg/kg) of either species.

An average uranium concentration of 0.002 mg/kg (wet weight) was reported for livers of fish representing Salmonidae, Coregonidae, Percidae, Cyprinidae, Osmeridae and Perciformes (Lucas *et al.* 1970).

Chou and Uthe (1995) measured the uranium concentrations in the digestive gland of lobster in the vicinity of an industrialized harbour (Belledune Harbour) in New Brunswick. The harbour is the site of a lead smelter, a fertilizer plant and a coal-fired powerstation. Uranium concentrations in digestive glands of lobsters were reported to range from 0.010 to 0.095 mg/kg (wet weight) with the vast majority (i.e., > 99.2%) of uranium as ^{238}U as opposed to ^{235}U.

Foran *et al.* (2004) reported uranium concentrations (along with other metals) in farmed Atlantic salmon and wild Pacific salmon available in Canada and the U.S. Farmed Atlantic salmon were purchased from commercial suppliers in Canada and the U.S. and included fish farmed in British Columbia, Maine, Chile and Norway. Wild Pacific salmon were from British Columbia, Washington and Alaska. Concentrations of uranium were not different in farmed Atlantic salmon versus wild Pacific salmon. Concentrations were less than 0.01 mg/kg (wet weight) (concentrations were only provided graphically and thus precise estimates of actual uranium concentrations were not available).

No significant accumulation of uranium occurred in snowshoe hare captured near the city of Elliot Lake, Ontario (Clulow *et al.* 1996). Total uranium concentrations were determined in hind leg bones by the neutron activation, delayed neutron, and gamma-counting techniques using a Slowpoke reactor. Concentrations were below detection (<0.4 mg/kg dry weight) in all the hares captured in 3 control areas (n=17) and at 3 mill tailing sites (n=12).

Commercial Foods

Estimates of dietary intakes of ^{238}U (the predominant U isotope) were available for Vietnam (Giang *et al.* 2001). Two methods were employed in the collection of food samples – market basket study and duplicate portion study – which included rice and grains, instants foods, potatoes, meat, fish, shrimp, eggs, beans, vegetables, fruits, sugar, fish sauces, and cakes. Uranium concentrations (as determined by ICP-MS) ranged from 0.36 to 9.32 ng/g (dry weight) in the food sampled. Daily intakes were estimated by region, ranging from 0.62 to 2.33 µg ^{238}U/day in the northern region, from 0.27 to 1.50 µg ^{238}U/day in the central region, and from 0.09 to 0.98 µg ^{238}U/day in the southern region of Vietnam.

Uranium concentrations in a typical Pakistani diet were recently reported (Akhter *et al.* 2003).

Food items (cereals, milk, root vegetables, pulses, meat, fish, eggs, oils, vegetables, fruits, sugar) were collected using the market basket method from 12 cities or districts in the country. Uranium concentrations (as determined by ICP-MS) ranged from 2.3 to 11 ng/g (dry weight) in the food sampled. Estimated daily intakes ranged from 1.4 to 6.7 µg U/day with a geometric mean daily intake value of 2.67 µg U/day.

A market basket study conducted in Japan reported a daily ^{238}U intake of 14 µBq/day (0.56 µg ^{238}U/day), based on the analysis of 336 foods in 18 food categories (Shiraishi *et al.* 2000). The highest food contributors to ^{238}U intake were marine products (50% from seaweed and 26% from fish and shellfishes).

The World Health Organization reported a worldwide daily intake of 1.1 µg ^{238}U/day (WHO 1996). Daily uranium intakes estimated for New York City, Chicago, and San Francisco were 1.3, 1.4, and 1.3 µg/day, respectively (Welford and Baird 1967). A dietary intake rate of 1.9 µg ^{238}U/day has been reported for Western Europe and North America (ICRP 1975). Vegetables and cereals were found to nominally contribute to the daily intake of uranium. Produce, especially root and stem vegetables, generally contained a 10-fold higher uranium concentration than animal produce (Berlin and Rudell 1979).

A daily intake (including food and water) of 4.4 $\pm$ 0.6 µg U/day was reported for Salt Lake City, Utah residents (Singh *et al.* 1990). The study did not evaluate uranium content in food, but inferred intake based on uranium measurements in the urine and feces of 12 subjects living in Salt Lake City. It should be noted drinking water samples in the study contained significant concentrations of uranium (*i.e.,* 1.46 $\pm$ 0.26 µg/L).

An average dietary intake value of 1.3 µg U/day was determined for New York City residents based on average daily intakes of 16.8 mBq/day, 0.69 mBq/day, and 14.7 mBq /day for ^{234}U, ^{235}U, and ^{238}U, respectively. These estimates were based on food concentrations determined in 1978. The mean concentration of total uranium on a weight basis for food samples from this study are summarized in Table 6. Shellfish contained the highest concentrations of uranium (total of 0.168 mg/kg). The authors reported that the consumption of animal and fish products contributed 41% of the total dietary intake determined for New York City residents.

Uranium-238 concentrations were reported in 20 food groups from the 2001 UK Total Diet Study and then used to estimate ^{238}U intakes for average and high intake consumers (UK FSA 2005). The highest UK total dietary exposures were determined for toddlers aged 1.5 to 2.5 years, ranging from 0.043 to 0.044 µg ^{238}U/kg bw/day for average intake consumers and from 0.077 to 0.079 µg ^{238}U/kg bw/day for high intake consumers. Adult dietary exposures ranged from 0.015 to 0.016 µg ^{238}U/kg bw/day (average intake) and from 0.026 to 0.028 µg ^{238}U/kg bw/day (high intake). The mean uranium contents reported in food samples from this study are summarized in Table 7. Fish and bread contained the greatest concentrations of ^{238}U (0.0035 mg U/kg for each). Cereals and bread contributed the most ^{238}U to the UK diet (*i.e.,* up to 65 % of the total intake for toddlers aged 1.5 to 2.5 years eating above average amounts of food).

Individual dietary intake values ranging from 0.3 to 19.5 µg U/day (averaged over 3 days) were

reported for adult residents of Ottawa, Ontario (Limson-Zamora *et al*, 2002). Uranium intakes via food were determined using the duplicate diet method; uranium concentrations in food were not reported. A median adult dietary intake of 1.02 µg U/day (0.014 µg U/kg bw/day, assuming an adult body weight of 70.7 kg) was determined from the data presented in this study (as recommended by one of the authors, B. Tracey, Health Canada, pers comm.). This intake is comparable to the average UK adult ^{238}U dietary intakes (*i.e.,* 0.015 to 0.016 µg ^{238}U/kg bw/day) (UK FSA 2005).

No data for uranium content in Canadian food were available and Canadian dietary intakes were only available for adults in a study by Limson-Zamora *et al.* (2002). For the purpose of this assessment, it was assumed that the ^{238}U concentrations determined in food for the 2001 UK Total Diet Study (Table 7) were representative of uranium levels in Canadian foods. This data was used to estimate Canadian dietary intakes.

The use of the UK data may be limited in that only ^{238}U (versus total uranium) concentrations were reported. However, the UK study reported a daily adult intake for ^{238}U (*i.e.,* 0.015 to 0.016 µg/kg bw/day) very similar to the median daily adult intake of total uranium (*i.e.,* 0.014 µg/kg bw/day) determined for Ottawa residents. In addition, various data (e.g., Thomas and Gates 1999; Chou *et al.* 1995) indicate that, when expressed on a weight basis, uranium in foods is most predominantly (>99%) in the form of ^{238}U. On this basis, the use of the UK food concentration data will likely not underestimate Canadian dietary exposure.

Existing Guidelines and Criteria in Various Media

In 1999 the Federal-Provincial-Territorial Committee on Drinking Water approved a revised Guideline for Canadian Drinking Water Quality for uranium. The interim maximum acceptable concentration (IMAC) is 0.02 mg/L (Health Canada 1999, 2002). The World Health Organization (WHO) has established a provisional drinking water quality guideline of 0.015 mg/L for chemical effects associated with naturally occurring uranium (WHO 2004a,b). This guideline is considered provisional due to limited information on health effects; in addition, the guideline value is below the level considered to be achievable through practical treatment methods and source protection.

The Ontario Drinking Water Standard for uranium (OME 2002) proposed a health related maximum acceptable concentration (MAC) of 0.02 mg/L. The BC Ministry of Environment Land and Parks (BCMELP 1995) recommended a maximum water concentration of 0.1 mg/L for drinking water and the protection of human health; however, this value was not included in subsequent versions of the Contaminated Sites Regulation, Schedule 10.

In 2000, the U.S. EPA established a drinking water standard for uranium of 0.03 mg/L under the Safety Drinking Water Act (Federal Register 2000).

The state of Arizona recommends an Ingestion Health-Based Guidance Level of 0.021 mg/L for groundwater (Buonicore 1995). Wyoming proposed 0.1 mg/L as a cleanup level for natural uranium in groundwater (Buonicore 1995).

Ontario's interim Provincial Water Quality Objective of 0.005 mg/L for uranium in surface water was proposed to protect aquatic life and recreational uses (OMEE 1999). The BCMELP (1995) recommends a maximum water concentration of 0.3 mg/L for the protection of freshwater aquatic life. Concentrations of 0.1 and 0.5 mg/L for minimal risk and hazard, respectively, were proposed for marine life (BCMELP 1995).

CCREM (1987) stated that the concentration of total uranium in irrigation water should not exceed 0.01 mg/L for continuous or intermittent use on all soils, or 0.1 mg/L for use up to 20 years on neutral and alkaline fine-textured soils. The BCMELP (1995) recommends a maximum water concentration of 0.2 mg/L for the protection of livestock health, 0.01 mg/L for irrigation water for continuous or intermittent use on all soil, and 0.1 mg/L for irrigation water for use up to 20 years on fine-textured soils with pH = 7.

Few American states and no Canadian agencies have cleanup standards or guidelines for uranium in soil or sediment. Arizona recommends an Ingestion Health-Based Guidance Level of 350 mg/kg of natural uranium in soil (Buonicore 1995).

Although not regulations, some regional offices of the U.S. EPA have published screening levels for contaminants to be used during site investigations under the Superfund Program. The U.S. EPA Region III presents two sets of risk-based concentrations (RBCs) for soluble uranium salts in soil. The first set is based on IRIS (EPA concensus, RfD last revised in 1989) and the RBCs for residential and industrial soils are 230 mg/kg and 3,100 mg/kg, respectively; the second set is more recent (circa 200X) but provisional for residential and industrial soils with values of 1.6 mg/kg and 200 mg/kg, respectively (U.S. EPA 2005). It is currently unclear whether these later values will replace those established in 1988. The U.S. EPA Region 9 has adopted Preliminary Remediation Goals (PRGs) for uranium that are based on the chemical toxicity only and for direct contact exposure pathways only. The concentrations for residential and industrial soils are exactly the same as the latter provisional values from Region III.

Efroymson *et al*. (1997) recommended a screening-level toxicological benchmark of 5 mg/kg soil for uranium for the protection of terrestrial plants, but confidence in the benchmark was low because it was based on a single study.

CHAPTER 3. ENVIRONMENTAL FATE AND BEHAVIOUR

Primary factors affecting the migration of uranium from deposits to the biosphere include groundwater leaching and physical-chemical conditions (Gordon 1992). Sandstone and permeable sedimentary rock are most susceptible to leaching by groundwater. Bedding planes and structural features (fractures and faults) in less permeable rocks also serve as conduits for groundwater transport. Chemical properties affecting uranium mobility include pH, pE, hydroxides, oxyhydroxides, oxides, clay minerals, organic matter, and complexing anions. These properties affect the adsorption, precipitation, and coprecipitation of uranium. Physical properties affecting uranium dispersion include the texture of sediment and soil (grain size and surface area), biota characteristics (type, organ, age, root distribution), and seasonal changes.

Atmosphere

Natural background levels of uranium in air are probably a result of re-suspension of soil and from volcanic eruption (ATSDR 1999; Kuroda *et al.* 1984). Uranium may be released to the atmosphere at various stages in the nuclear fuel cycle, including mining, milling, refining, fuel fabrication and fuel reprocessing (Tracy and Meyerhof 1987). Production of phosphate fertilizers using phosphate rocks containing uranium can also be a source of uranium in the air (NCRP 1984).

Uranium in the air is probably largely present as particulate matter (PM) (ATSDR 1999). Approximatley 70% of atmospheric uranium was reported to be present in PM >2 µm in diameter, with 40-50% >7 µm in diameter (Sugiyama *et al.* 1988), suggesting that uranium may be more associated with coarse (PM_{10}) versus fine ($PM_{2.5}$) particulates. Transport of uranium-bearing particles from air to water, plants, or soil occurs by wet and dry deposition (ATSDR 1999). The particles possess the same physical and aerodynamic properties as general urban dust, and their atmospheric transport will depend on the particle size, distribution, and density (Metzger *et al.* 1980; Tracy and Meyerhof 1987; ATSDR 1999).

Little information exists regarding the abiotic transformation and degradation of uranium and its compounds in the air. The lifetime of the airborne hydrolysis product of uranium hexafluoride ($UO_2F_2{\cdot}F_2{\cdot}nH_2O$) is about 35 minutes (Bostick *et al.* 1985).

Soil

Uranium is mobilized from rock by the weathering of uraninite (UO_2). The action of surface waters and groundwater causes oxidative dissolution of uraninite to the soluble uranyl ion (UO_2^{2+}). Worldwide, from 27 000 to 32 000 t uranium are released from igneous, shale, sandstone, and limestone rocks annually by weathering and natural erosion (Eriksson 1960; Bowen 1966; Environment Canada 1983). The major anthropogenic activities that release uranium into the soil are uranium mining and milling, uranium processing, phosphate mining, heavy metal mining, coal use, and inappropriate waste disposal (ATSDR 1999).

In soil, the U^{4+} valence state (typically solid UO_2) of uranium occurs in strongly-reducing environments and is formed by the oxidation of organic matter or iron in the soil. Tetravalent uranium forms hydroxides, hydrated fluorides, and phosphates which are strongly adsorbed and very immobile in soils. The U^{6+} valence occurs in oxidizing environments (UO_2^{2+}) and is strongly adsorbed by soils, forming stable complexes with many ligands - notably carbonates - and organic complexants. High ligand concentrations can result in a lower positive or negative charge and increase mobility of the complexed uranium (Sheppard and Evenden 1987).

Soil properties that affect uranium mobility (and subsequent uptake by biota) include aeration (water saturation, high biological or chemical oxygen demand), carbonate content (organic material content, pH, parent material, weathering), and cation exchange capacity (texture, clay content, organic matter, pH). A higher soil cation-exchange-capacity will retain more uranium, while carbonate in the soil increases the mobility of uranium through the formation of anionic U and CO_3 complexes (Allard *et al.* 1982; Sheppard and Evenden 1987). Uranium does not migrate substantially in loam compared to sandy soils (Sheppard *et al.* 1984). Uranium migration in soil occurs over the period of a few months, depending on sorption, and may be upwards when there is a net water deficit or downwards as a result of net leaching (Sheppard *et al.* 1984).

Soil properties reported to increase mobility and plant accumulation of uranium include acidic soils with low adsorptive potential, alkaline soils with carbonate minerals, and the presence of chelates (citric acid) (Shahandeh and Hossner 2002a).

Transfer coefficients, or concentration ratios (CR), were reported by Gordon (1992) as ratios of the maximum concentrations determined for soil/groundwater at uranium ore bodies in B.C. and Ontario. Soil-groundwater CR values of 4,588 and 1,507 were reported for the Blizzard (B.C.) deposit and the South March (Ontario) deposit, respectively.

Microorganisms can degrade soluble organo-uranium compounds in soil and rocks using ligands as a source of carbon and energy which enhances uranium precipitation and deposition. Fungi (*Aspergillus ochraceus* and *Penicillium funiculosum*) were able to take up large amounts of soluble uranium in their mycelium from rocks. Uranium uptake was increased by nutrient deficiency, higher uranium content in the rock, and lower pH (Berthelin and Munier-Lamy 1983).

Water and Sediment

The release of uranium to the water occurs primarily from production of uranium from ore, from the use of uranium in nuclear reactors, and from disposal of solid wastes from these industries (ATSDR 1999). The most important factors controlling the mobility of uranium from sediment to the water phase are oxidation-reduction potential, pH, characteristics of complexing agents or ligands, and the nature of sorbing materials in water (ATSDR 1999).

The uranyl ion (UO_2^{2+}) forms stable salts and complexes with many commonly occurring anions in aquatic environments (CCREM 1987). The chemical speciation of uranium ions in aqueous

solution is quite complex because of the many possibilities of complexing reactions with most other ions (CCREM 1987). Hydrolytic reactions may also lead to polymeric ions (CCREM 1987). Uranium may be precipitated as insoluble UO_2 or adsorbed by clays and hydrous metal oxides (Cotton and Wilkinson 1980).

In aerobic waters, the most significant complexing agent for uranium is carbonate (CCREM 1987). Below pH 5, the predominant uranium species are UO_2^{2+} and UO_2OH^+; however, above pH 5 and in the presence of carbon dioxide, carbonate species predominate (CCREM 1987). At pH 5, the dominant form of uranium is neutral UO_2CO_3; at pH 6-7, $UO_2(CO_3)_2^-$ dominates; and at pH 8, $UO_2(CO_3)_2^{4-}$ dominates (Halbach *et al.* 1980; CCREM 1987). These uranyl carbonate species are all quite stable in the typical ranges of redox potential found in natural water (CCREM 1987). Sulphur complexes are also soluble, whereas potassium and phosphate complexes are quite insoluble (Taylor 1979; Giblin *et al.* 1981). Bog waters transport uranium mainly as dissolved uranyl fulvate, but a steady feed of rainwater prevents a higher uranium concentration in the aqueous environment (Halbach *et al.* 1980).

Sorption plays a dominant role in determining the fate of uranium in the aquatic environment (CCREM 1987). Sorption to clay minerals such as kaolinite below pH 5 and sorption to hydrous ferric oxide at higher pH in aerobic waters will reduce the mobility of uranium (Giblin *et al.* 1981). The transport of uranium from water to organisms occurs primarily through the sediment, which usually contains higher levels of uranium than water (ATSDR 1999). Aquatic microorganisms reduce soluble, oxidized forms of uranium to insoluble forms (Lovley *et al.* 1991).

Transfer coefficients, or concentration ratios (CR) were reported by Gordon (1992) as ratios of the maximum concentrations determined for sediment/groundwater and sediment/surface water at uranium ore bodies in B.C., Saskatchewan, and Ontario. Sediment-groundwater transfer coefficients of 254 and 151 were reported for the Blizzard uranium deposit (B.C.) and the South March uranium deposit (Ontario), respectively. Sediment-surface water transfer coefficients of 12,222 and 12,705 were reported for the Bancroft deposit (Ontario) and the Blizzard deposit respectively.

Biota

As discussed below, low levels of bioconcentration and bioaccumulation of uranium have been observed in various species (i.e., direct uptake by organisms from the medium in which they live and/or through consumption of food containing uranium). However, there does not appear to be any evidence that uranium biomagnifies, i.e., tissue concentrations of accumulated uranium do not increase from one trophic level to the next.

Numerous studies have looked at the uptake and accumulation of uranium by plants. The mobility of uranium in plant tissues is limited, as it tends to adsorb on cell wall materials; therefore, concentrations are typically higher in tissues found lower on the plant and are highest on the root surfaces (Sheppard and Evenden 1988b; Shahandeh and Hossner 2002). Shahandeh and Hossner (2002) grew various plant species in soil spiked with 100 mg U/kg (added as uranyl

nitrate) to determine how much uranium was bioaccumulated. Concentrations of uranium measured in the shoots of the various plant species ranged from 3.2 to 24 mg U/kg. Uranium concentrations in the roots were 30-50 times greater than in the shoots, ranging from 89 to 810 mg U/kg of dry root weight. Generally, dicotyledenous plants accumulated more uranium than monocotyledonous plant species (grasses). For example, in both shoots and roots, sunflower (*Helianthus annus*) and Indian mustard (*Brassica juncea*) accumulated the highest concentrations of uranium, while wheat (*Triticum aestivum*) and ryegrass contained the lowest concentrations (Shahandeh and Hossner 2002). When sunflower was grown in several different soil types contaminated with uranium, greater accumulation of uranium was observed in the tissues of those plants grown in calcareous soils than those in acid soils (Shahandeh and Hossner 2002).

Uptake of uranium by plants may be influenced by pH. Accumulation of uranium in shoots of peas (*Pisum sativum*) grown in nutrient solution at pH 6.0 and pH 8.0 was less than 20% and 5%, respectively than uptake at pH 5.0 (Ebbs et al. 1998). At pH 5.0 the uranium was present primarily as the free uranyl cation which may be more readily taken up and translocated than other uranium species (Ebbs et al. 1998). Uptake of uranium was also affected by the presence of phophorus. When grown in nutrient solution containing 5 μM uranium, in the absence of phosphorus, growth of both shoots and roots of pea plants was severely inhibited compared to the controls (Ebbs et al. 1998). However, when the plants were exposed to both uranium and phosphorus, the root and shoot lengths were not significantly different from control plants. The authors suggested that complexation of the uranium with phosphate may have reduced its bioavailability and its toxic effects (Ebbs *et al.* 1998).

It has been speculated that, in addition to uptake from soil, atmospheric deposition may be a significant uranium exposure pathway for plants (Sheppard *et al.* 1989).

The ability of plants to accumulate substances from soil is often evaluated by calculating a concentration ratio. A concentration ratio is the concentration of U in dry tissue of an organism divided by the total concentration of U in the dry soil (Sheppard and Evenden 1992a). In a review of plant concentration ratios reported in the literature, Sheppard and Evenden (1988b) found that mean concentration ratios for uranium can range over 30,000-fold. They noted that fine soils resulted in significantly lower concentration ratios than coarse, peat, or tailings soils. They also found that root crops had higher concentration ratios than fruit, cereal, shrub or leafy vegetable crops (Sheppard and Evenden 1988b). In study that looked at uranium accumulation in earthworms, beans, and radish for several different soils, higher concentration ratios were seen in the sandy soils (Sheppard and Evenden 1992a). Concentration factors between roots of birches and aspens and uranium mine tailings were below unity; however, concentration factors calculated between roots and soils were at or above unity (Environment Canada 1988). The ratios between stems and leaves reflected the consistently higher concentrations of uranium in the leaves of the trees growing on both tailings and control sites; the ratios also suggested species-specific translocation within these plants (Environment Canada 1988).

Uranium concentrations in plants have been shown to be linearly related to uranium concentrations in soil pore water, however, soil-plant systems are more complex and CR values vary according to measurement and reporting techniques, plant types, soil properties, and

climate. Sheppard and Evenden (1992a) noted a curvilinear relationship between plant concentration ratios and concentrations of uranium in the soil, suggesting that a single concentration ratio may not be representative over all soil uranium concentrations. Plant-soil CR values can vary between 3 to 5 orders of magnitude; they are lognormally distributed and, ideally, averages should be computed as geometric means (Sheppard and Evenden 1987).

Sheppard and Evenden (1987) determined plant-soil CR values (dry weight) for a variety of plant types, including cereals (0.001 to 0.0083), fruits (0.0002 to 0.01), vegetables (0.0002 to 0.008), and root crops (0.002 to 0.007). Bean-soil CR values in various soils ranged from 0.035 to 0.066. Radish-soil CR values ranged from 0.013 to 0.237. Higher CR values were reported for sandy soils (Sheppard and Evenden (1992a). Farm vegetable-soil CR values of the order of 10^{-3} were reported by Lakshmanan and Venkateswarlu (1988). An overall geometric mean plant-soil CR value of 0.013 (dry weight) was recommended for uranium by Sheppard *et al.* (1989) based on 12 different soil (100 mg U/kg) and crop (field and garden) combinations.

Bioaccumulation coefficients (CRs) were reported for roots and shoots of plants (dicotyledon, monocotyledon, and *Brassica* sp. plants) grown in soil containing 100 µg U/g (Shahandeh and Hossner 2002b). Uranium was significantly more concentrated in the roots (89 to 810 mg U/kg, dry weight) versus the shoots (3.2 to 24 mg U/kg, dry weight) for all plant species evaluated. Plant shoot-soil CR values ranged from 0.03 (wheat) to 0.38 (sunflower), plant root-soil CR values ranged from 0.3 (wheat) to 8.1 (sunflower).

Plant-soil CR values from 0.026 to 4.5 (dry weight basis) were reported for uranium based on soil and vegetation (black spruce needles and twigs, jackpine needles and twigs, leatherleaf, Labrador tea, and blueberries) samples collected from a control site in northern Saskatchewan (Thomas 2000). The author reported an overall arithmetic mean plant-soil CR value of 0.51, based on a uranium soil concentration of 1 mg/kg.

Uranium accumulation was reported in plant shoots exposed to uranium (750 mg/kg) contaminated soil (Huang *et al.* 1998). India mustard (*Brassica juncea*) and Chinese mustard (*Brassica narinosa*) plants grown in the uranium soil accumulated 5 mg U/kg in their shoots. The addition of citric acid to soils increased desorption of uranium from soil to soil solution and triggered a rapid accumulation of uranium by the plants; a 1000-fold increase in shoot uptake (*i.e.,* 5000 mg U/kg) was reported for India and Chinese mustard plants exposed to citric acid treated soil containing 750 mg U/kg.

Uranium uptake by barley plants was evaluated using soil collected from the Prairie Flats surficial uranium deposit near Summerland B.C. (Van Netten and Morley 1982b). Vegetation-soil concentration ratios (dry weight basis) were low, and tended to decrease with increasing soil concentrations, from a CR = 0.433 reported for soil containing 3 mg U/kg (pH 7.37) to a CR = 0.032 for soil containing 313 mg U/kg (pH 7.55). The results of this study indicate that barley plants do not bioaccumulate uranium from soils at this location.

Uranium uptake was evaluated for plants (lettuce, tomato, squash, and radish) grown in control soil (2.3 $\pm$ 3.0 mg U/kg) and irrigated with well water from the Nambe region of northern New

Mexico, which contained uranium at <1 (control), 150, 500, or 1200 µg/L (Hakonson-Hayes *et al*. 2002). Increasing uranium concentrations in water resulted in a linear increase in uranium concentrations in plants (collected at maturation). Mean uranium concentrations (dry weight) in edible plant tissue ranged from 8 to 67 mg/kg in tomato, from 13 to 285 mg/kg in squash, from 82 to 2879 mg/kg in radish, and from 79 to 2304 mg/kg in lettuce. Baseline plant-soil CR values (uranium uptake from control soil irrigated with <1 µg U/L) ranged from 10^{-3} to 10^{-2} for edible tissue. Plant-soil CR values representing uptake from irrigation treatment and corrected for background uranium concentrations, ranged from 10^{-2} to 1.6 for edible plant portions. An implication of this study was that the primary source of plant-available uranium was water, versus soil-bound uranium.

Transfer coefficients (CRs) were calculated by Gordon (1992) as ratios of the maximum concentrations determined for plant/soil, plant/groundwater, and plant/surface water, at uranium ore bodies in B.C., Saskatchewan, and Ontario. All plant concentrations were reported as ash weights. Plant-soil transfer CR values of 5.3 and 0.1 were reported for the Bancroft uranium deposit (Ontario), 1.1 for the Blizzard deposit (B.C.), 0.029 for the Midwest Athabasca deposit (Saskatchewan), and 0.0016 for the South March deposit (Ontario). Plant-groundwater CR values of 4,941 and 24, 706 were reported for the region of the Blizzard uranium ore deposit (B.C.), while a plant-groundwater CR value of 2.5 was reported for the South March uranium ore deposit (Ontario). Plant-surface water CR values of 247,059 and 1,235,294 were determined for the Blizzard uranium deposit (B.C.), while values of 71 and 367 were reported for the Bancroft (Ontario) and Midwest Athabasca (Saskatchewan) deposits, respectively.

Plant-water bioaccumulation coefficients (CRs) were reported in a pilot-scale rhizofiltration experiment which exposed sunflower plants (*Helianthus annuus*) to 750 ml of water containing uranium at a concentration of 56 µg/L (Dushenkov *et al*.1997). The sunflower plants removed >95% of the uranium in solution within 24 h. Uranium in the sunflower was concentrated in the roots (>99%), with negligible transport to shoots. Further testing indicated a linear increase in root concentration with increasing uranium water concentrations, with more uranium adsorbed at pH 5 compared to pH 7. Average plant-water CR values based on uranium concentration in sunflower roots (dry weight) versus uranium concentrations in water (ranging from 10 to 2430 µg/L) were 6624 ($\pm$870) and 3379 ($\pm$430) for pH 5 and pH 7, respectively.

Plant uptake of uranium in irrigation groundwater was studied in a lab setting using a root crop (carrots), a fruit (squash), and leafy forage (Sudan grass) grown in control soil (Baumgartner at al. 1999). Irrigation water contained from 20 to 5000 µg U/L. Squash did not accumulate uranium at any dose tested. Uranium uptake by Sudan grass was uneven (poor correlation with water concentrations) and carrots irrigated with water containing 5000 µg U/L accumulated less than 8 mg U/kg (dry weight).

Tracy *et al.* (1983) reported that uranium in gardens of Port Hope, Ontario, could accumulate in garden produce. In this analysis, concentrations of uranium were measured in soils and plants from gardens impacted by uranium processing wastes. Soil concentations ranging from 2.3 to 760 mg/kg of uranium were reported. The greatest concentrations were found in root and stem vegetables (concentrations up to 180 mg/kg wet weight were reported in carrots); however, raspberries were also noted to have elevated concentrations. A geometric mean soil to plant concentration factor of 75 for uranium was reported by Tracy *et al.* (1983).

Microorganisms appear to act as sinks for uranium, which is accumulated and concentrated to high levels in cell walls (Berthelin and Munier-Lamy 1993). Of ten aquatic fungal cultures tested, five (*Alternaria tenulis*, *Chaetomium distortium*, *Fusarium* sp., *Saccharomyces cerevisiae* and *Trichoderma horzianum*) were capable of biosorbing more than 90% of uranium present in aqueous solution at an initial concentration of 150 mg U/L (Khalid *et al.* 1993). Biosorption of the uranium was highly dependent on solution pH, with optimal uptake of the uranyl acetate occurring at pH 5.5. (Khalid *et al*. 1993). *Streptomyces sp.*, either living or dead, are able to accumulate UO_2^{2+} ions, which binds to cell wall sites as well as to cytoplasmic structures within the cells of the bacteria (Golab *et al.* 1991). The uptake of uranium was examined for indigenous bacterial species (*Bacillus*) occurring in a uranium waste pile in Saxony, Germany (Selenska-Pobell *et al.* 1999). The study demonstrated that bacteria species *B. spaericus, B. cereus,* and *B. megaterium* and their spores selectively accumulated uranium from contaminated waters. In vegetative cells, sorption of approximately 90% of the uranium (present in the water at 72.1 µg U/L) was observed, while spores tended to show slightly lower sorption with more variability across different *Bacillus* strains (Selenska-Pobell *et al.* 1999). Furthermore, attempts to extract uranium from some of the spores of the uranium waste isolate (using EDTA) failed, suggesting irreversible binding of U by the spores.

A few studies have also looked at bioaccumulation of uranium in soil invertebrates and mammals. Soil-to-earthworm concentration ratios for various soil types were reported to range from 0.082 to 2.38 (Sheppard and Evenden 1992a). The internal tissues from two sympatric small mammal species (deer mouse and valley pocket gopher) had soil-to-tissue ratios of 10^{-3} to 10^{-4}, although gastrointestinal contents of these mammals had soil-to-tissue ratios of greater than 10^{-1} (i.e., mean uranium levels in the GI were more than 10% of the uranium concentration measured in the soil) (Miera et al. 1980). Clulow et al. (1996) measured the concentrations of various radionuclides in snowshoe hare (*Lepus americanus*) living in the immediate vicinity of revegetated uranium tailings. Total uranium concentrations in hind leg bones of the hares were below the detection limit of 0.4 mg/kg dry weight. The authors did, however, measure elevated levels in the hares of radium-226, a radioactive decay product of uranium-238 (Clulow et al. 1996).
The ability of biota to restrict uranium uptake might be impaired by sublethal toxicity, however, uranium does not appear to bioaccumulate in vegetation or soil invertebrates to a significant degree.

A mean biota-sediment accumulation factor of 0.007 (from 0.004 to 0.008, dry weight basis) was calculated for fathead minnows (*Pimephales promelas*) exposed to uranium in lake sediments

(Port Hope Harbour, Ontario) (Senes Golder Partnership (SGP) 2003). Tissue residues were low relative to the high sediment U concentrations tested (960 to 3620 mg/kg). The authors also reported that the accumulation factor did not increase at higher sediment concentrations, suggesting a control of the release of uranium from the sediments.

Bioconcentration factors (biota-water) ranging from 0.1 to 0.3 (wet weight basis) were reported for total uranium in fish (whitefish and white sucker) caught in northern lakes (Muir *et al.* 1992).

Uranium does not biomagnify in the aquatic food chain, rather, concentrations in biota decline with each successive trophic level of aquatic species (ATSDR 1999). Concentration ratios (CR) were developed to estimate food-chain transfer based on ratios of average caribou tissue (bone, liver, kidney, muscle) concentrations to rumen content concentrations (Thomas and Gates 1999). The CR values indicated transfer rates of 16 to 29% (CR = 0.16 to 0.29) for uranium from the rumen to the bone, liver, kidney, or muscle tissue (wet weight basis). CR values reported on a dry weight basis indicated transfer rates of 11 to 20% (CR = 0.11 to 0.20). These CR values may overestimate uranium intake by caribou as all uranium concentrations in rumen contents were below analytical detection (i.e., <20 ppb or 0.02 mg/kg) and one-half the detection limit was assumed. CR values estimated based on uranium content in the food source (i.e., lichen) rather than in the rumen contents were lower, resulting in transfer estimates of 2.7 to 4.8% (CR = 0.027 to 0.048).

Table 8 lists concentration ratios that have been reported for various species (or were calculated based on concentration data provided in the studies). Overall, uranium does not appear to bioaccumulate in vegetation or soil invertebrates to a significant degree, as mean concentration ratios for species, in most cases, are less than 1.

CHAPTER 4. BEHAVIOUR AND EFFECTS IN BIOTA

Most of the studies on uranium toxicity described below were conducted by adding uranium salts, such as uranyl nitrate, to the soil. However, there are also a few studies that involved the use of depleted uranium metal. At present, depleted uranium is not likely to be found in Canadian soils as it is not used in Canadian munitions. Depleted uranium can, however, be found internationally in soil at sites where weapons containing depleted uranium have been tested, such as several military proving grounds in the United States, or used in battle, such as in Iraq. Studies with depleted uranium may tend to show lower levels of toxicity as metallic forms of uranium will have low bioavailability.

Mode of Toxicity

Uranium can produce toxic effects through both chemical toxicity and radiotoxicity. Radiotoxicity can result from uranium incorporated into an organism's tissue that is emitting radiation (internal dose), as well as from uranium that is emitting radiation adjacent to the organism (external dose) (Environment Canada – Health Canada 2003). Due to the long half-life and slow rate of decay of uranium isotopes, their radiotoxicity is expected to be relatively low. Therefore, the adverse effects observed in the following studies on uranium toxicity can be assumed to result largely from chemical toxicity, rather than radiation effects.

Soil Microbial Processes

Uranium is not highly toxic to microorganisms relative to other heavy metals such as cadmium (Berthelin and Munier-Lamy 1993). Sheppard *et al.* (1992) reported significant depression of phosphatase activity in 3 of 11 different soils at a concentration of 1000 mg/kg, with no adverse effects observed at 100 mg/kg (Table 9).

Meyer *et al.* (1998a) examined the effects of depleted uranium on soil microbial processes. The depleted uranium was in the form of schoepite [$UO_2(OH)_2 \cdot H_2O$] collected from deployed munitions, and was finely ground prior to addition to the soil (Meyer *et al*. 1998a). The authors found that a uranium concentration as low as 500 mg/kg resulted in decreased soil respiration rates. At a uranium concentration of 25 000 mg/kg, decomposition processes were inhibited, but there was no effect on nitrogen mineralization (Meyer *et al.* 1998a). The applicability of these results to conditions found in Canada is limited as depleted uranium is not likely to be found in Canadian soil; nonetheless, the mineral schoepite does occur naturally in Canada (Chatterjee 1977). It should also be noted that this form of uranium is not very soluble and consequently might have low bioavailability.

No other literature on the toxicity of uranium to soil microbial processes was identified for this review. There are, however, studies that have looked at toxicity of uranium to aerobic bacteria commonly found in sewage treatment plants. In a 66-hour study involving addition of uranyl nitrate hexahydrate [$UO_2(NO_3)_2$ x $6H_2O$] to culture medium, the growth of bacterium *Zoogloea ramigera* was shown to be completely inhibited at 1000 mg/L, and the lag phase was increased by 20 hours compared to the controls by concentrations as low as 1 mg/L (Norberg and Molin

1983). Tuovinen and Kelly (1974) examined the effects of uranyl sulphate on microbial processes of the bacteria *Thiobacillus ferrooxidans* grown in an aqueous culture medium. Inhibition of ferrous iron oxidation was observed at concentrations as low as 0.4 mM (i.e., 95 mg/L). Inhibition of carbon dioxide fixation in the bacteria cultures was also observed to occur with exposure to uranium (Tuovinen and Kelly 1974). When uranium concentrations were increased gradually over time, increased tolerance was observed, apparently resulting from the selection of tolerant mutants, rather than from adaptation of the whole population (Tuovinen and Kelly 1974).

A summary of all toxicity studies that were reviewed on microbial process is provided in Table 9.

Terrestrial Plants

There is some evidence to suggest that hormesis (i.e., stimulation of growth at low concentrations) may occur in some plants exposed to uranium. In a study that examined effects of depleted uranium on growth of three grass species, one of the species showed evidence of hormesis (Meyer *et al*. 1998b). In a study by Gulati *et al*. (1980), wheat exposed to uranium concentrations of 1.5, 3.0, and 6.0 mg/kg showed increased yield compared to controls. This trend was not observed with tomato, however, where even the lowest concentration, 1.5 mg/kg, resulted in a 10% reduction in yield (Gulati *et al*. 1980). Nonetheless, observed growth increases in certain plants exposed to uranium-rich soils have lead some researchers to suggest that uranium may be a micronutrient for higher plants (Cannon 1952; Morishima *et al.* 1976); however, conclusive proof is lacking. Meyer *et al*. (1998b) suggest that a potential mechanism for the hormesis might be enhanced uptake of phosphorus due to interactions of uranium with phosphate to form complexes (Meyer *et al*. 1998b).

In plant tissue, uranium probably exists as the uranyl ion in the water-soluble fraction, or is bound to cell wall proteins (Whitehead *et al.* 1971). The phytotoxicological data available for uranium are presented in Table 10. Soil conditions, such as pH, are provided in this table. The mechanisms of phytotoxicity involve inhibition of enzyme systems and binding to nucleic acids (Feldman *et al.* 1967). Sheppard *et al.* (1983) and Sheppard and Thibault (1981) concluded that the toxic effects of uranium are the result of chemical toxicity rather than radiation-related toxicity based on minimal radiation measured in soil and air around the experimental plants.

Sheppard *et al.* (1984) observed no significant effects of uranium at 100 mg/kg soil on shoot yields of alfalfa and Swiss chard. Also at a uranium concentration of 100 mg/kg, no effects were observed on the mortality of Scots pine (*Pinus sylvestris*) seedlings (Sheppard *et al.* 1985). Mortality of blueberry plants (*Vaccinium angustifolium*) was found to occur at a uranium concentration of 10 000 mg/kg, but no adverse effects were observed at 8 000 mg/kg (Sheppard and Evenden 1988a).

Sheppard *et al.* (1992) attempted to identify the phytotoxic threshold of soil uranium by studying six plant species. Beans (*Phaseolus vulgaris*) were grown to maturity in an outdoor lysimeter study with limed boreal forest and garden soils. Uranium levels of 1000 mg/kg did not affect the

emergence of the beans (no observable effects concentration (NOEC) >1000 mg/kg); but was detrimental to seedling weight. In the same study, crops of corn (*Zea mays*), lettuce (*Lactuca sativa*), tomato (*Lycopersicon esculentum*), *Brassica rapa* and white pine (*Pinus strobus*) exposed to 1000 mg/kg uranium exhibited significantly reduced germination (Sheppard *et al.* 1992). In the limed boreal soil, all crops had significantly reduced germination at 1000 mg/kg of uranium. In addition, the rate of germination of pine and the dry weight of corn seedlings were also significantly decreased by this uranium concentration. In the garden soil, the authors observed lower toxicity and attributed it to more organic matter and a finer texture. There were no systematic decreases in germination up to 1000 mg/kg and as high as 10 000 mg/kg for *B. rapa*. None of the measurements in the mature *B. rapa* indicated detrimental effects of uranium below 300 mg/kg.

Sheppard *et al*. (2004) examined the effects of uranium on northern wheatgrass (*Elymus lanceolatus*) through exposures in both a natural loam soil collected from Port Hope and a limed sand soil. The plants were exposed to two uranium concentrations plus a control in the two types of soil, as well as being grown in an articial reference soil. In the Port Hope soil, they reported 51d NOEC and LOEC values of 838 and 3190 mg U/kg, respectively, for shoot length, root length and whole plant dry weight. No effects on emergence of northern wheatgrass in the Port Hope soil were observed at the highest tested concentration of 3190 mg U/kg. In the limed sand soil, NOEC and LOEC values of 994 and 2580 mg U/kg, respectively, were reported for root length. No effects were observed at the highest tested concentration of 2580 mg U/kg for emergence, shoot length and whole plant dry weight of northern wheatgrass grown in the limed sand soil.

Meyer and McLendon (1997) studied the effect of depleted uranium on three grass species. The depleted uranium was in the form of the mineral schoepite and was obtained by grinding up weathered material from deployed munitions. The authors demonstrated that depleted uranium was relatively non-toxic to grasses (buffalograss, *Buchloe dactyloides*; little bluestem, *Schizachyrium scoparium*; and purple threeawn, *Aristida purpurea*). Decreases in plant biomass, fecundity, and long-term survivability were observed only at the highest uranium level (25 000 mg/kg). The low observed toxicity may reflect low bioavailability of the uranium due to the low solubility of schoepite.

In contrast to the studies above, Aery and Jain (1998) observed adverse effects in wheat at very low concentrations of uranium in soil. Spike number, seed number and seed weight were significantly inhibited at the lowest concentration tested, 1 mg/kg soil. An inhibition in root elongation, root biomass, and shoot biomass (less pronounced) were also observed with increasing uranium concentration in the soil. Aery and Jain (1998) attributed the decreases to a reduced extensibility of the cell wall, decreased cellular turgor, and/or lower mitotic activity in the meristematic zone. The fact that Aery and Jain (1998) observed adverse effects at uranium concentrations that were as much as two orders of magnitude lower than the effects concentrations reported by many other researchers, suggests that these results should be considered with caution.

Effects of uranium on wheat were also studied by Gulati *et al*. (1980). They exposed the plants in a sandy loam soil spiked with uranyl nitrate. At the highest concentration tested, 6 mg/kg, no adverse effects were observed on the wheat yield, and in fact, yield was higher than in controls. Tomato plants, on the other hand showed greater sensitivity, with decreased yield at a uranium concentration of 3.0 mg/kg (Gulati et al. 1980).

A summary of all terrestrial plant toxicity studies that were reviewed is included in Table 10.

Terrestrial Invertebrates

Sheppard *et al.* (1992) attempted to determine the toxicity threshold of uranium to earthworm survival and growth (Table 11). In a limed boreal forest soil, worms in 1000 mg/kg were distinctly smaller and more darkly coloured at 14 days than worms in the control soil, and did not survive 75 days. The authors had some difficulty interpreting the data, but concluded that only the low survival at 1000 mg/kg in the limed boreal forest soil was an effect of uranium. The authors suggested growth rate or reproductive success would be more sensitive indicators of earthworm toxicity to uranium.

Sheppard *et al.* (2004) examined the toxicity of uranium to the earthworm *Eisenia andrei*. Assays were conducted with three different soil types, a natural loam soil collected from Port Hope, a fine sandy-loam "garden soil" enriched in organic matter, and a limed sand soil. In each of these assays, the earthworms were exposed to nominal concentrations of approximately 0, 1000, and 3000 mg U/kg as well as a reference artificial soil. Analyses of the soils were conducted to determine actual measured concentrations in each of the soils. The authors reported 14d NOEC values of >838, >994, and >1120 mg U/kg for survival in the Port Hope, garden, and sand soils, respectively. They also reported 56d NOEC values of >838, >994, and >1120 mg U/kg in the three soils for number of juveniles, number of hatched cocoons, number of unhatched cocoons and juvenile wet mass (Sheppard *et al*. 2004).

Sheppard *et al*. (2004) also examined the toxicity of uranium to two species of collembolans, *Onychiurus folsomi* and *Folsomia candida*. Initial studies with *O. folsomi* were conducted over a 35d exposure period with a natural loam soil from Port Hope and a limed sand soil, each containing nominal concentrations of approximately 0, 1000, and 3000 mg U/kg. With the Port Hope soil, a NOEC of 838 mg U/kg and a LOEC of 3190 mg U/kg were observed for adult survival, number of juveniles, and fecundity (number of juveniles per surviving adult). For the limed sand soil, NOEC and LOEC values of 1 and 994 mg U/kg, respectively, were reported for adult survival and fecundity. NOEC and LOEC values of 994 and 2580 mg U/kg, respectively, were reported for number of juveniles in *O. folsomi* exposed in the limed sand soil. A second set of assays was conducted with both *O. folsomi* and *F. candida* exposed to a series of 5 to 7 uranium concentrations in various soil types. For *O. folsomi*, 35d EC_{20} values for adult survival ranged from 92 to 480 mg U/kg in the various soils. *F. candida* tended to be less sensitive, with 28d EC_{20} values for adult survival ranging from 350 to 1030 mg U/kg in various soil types. These results should be treated with caution as survival of both species in one of the control soils was very poor. For both species, Sheppard *et al*. (2004) also determined effects on reproduction by counting number of juveniles over a 35d exposure for *O. folsomi* and a 28d exposure for *F.*

candida. The EC_{20} values for reproduction in *O. folsomi* and *F. candida* were 150-1030 and 840-2200 mg U/kg, respectively.
Several studies have been conducted with exposure of earthworms to uranium in solution on filter paper. Ribera *et al*. (1996) exposed the earthworm *Eisenia fetida andrei* to a uranium acetate salt solution and reported a 96h LC_{50} of 40 µg U/cm^2 (95% confidence interval of 0 - 123 µg/cm^2). In a similar study, Labrot *et al*. (1999) reported a 96h LC_{50} of 13.5 µg U/cm^2 (95% C.I. = 2.6 – 22.9 µg U/cm^2) for *Eisenia fetida andrei* exposed to uranyl acetate. Labrot *et al*. (1996) also exposed *E. fetida andrei* to a uranyl acetate solution and examined various biochemical effects. Significant decreases in acetylcholinesterase activities, a biomarker of neurotoxicity, were observed at uranium concentrations as low as 0.5 µg/cm^2. Decreased levels of malondialdehyde, a biomarker of oxidative stress, were observed at a concentration of 1.0 µg U/cm^2. Two biomarkers of hydroperoxide detoxication (catalase and glutathione peroxidase) did not show a clear dose-response relationship at the concentrations tested (Labrot et al. 1996). Again, it should be noted that these studies did not involve exposure through soil.

A summary of all invertebrate toxicity studies that were reviewed is included in Table 11.

Livestock and Wildlife

Uranium has not been demonstrated to be essential in animals (NRC 1980). Uranium is of low toxicity to birds in comparison to the toxicity of other heavy metals such as lead. The 7-d LC_{50} of uranium nitrate for 4-wk-old Leghorn cockerals has been reported to be 235 mg/kg bw (body weight), while the 7-d LOEL for death was 160 mg/kg bw; abnormal biochemical measurements and renal and hepatic lesions persisted for 72 hours following a single subcutaneous dose of uranium nitrate (Harvey *et al.* 1986). Haseltine and Sileo (1983) noted that uranium did not affect mortality or body, kidney and liver weights of 9-month-old American black ducks (*Anas rubripes*) fed uranium for 6 weeks at concentrations as high as 1600 mg/kg in the diet. Kupsh *et al.* (1991) observed degeneration and necrosis of tubular epithelium in the distal tubules in the kidneys of mature male Japanese quail (*Coturnix coturnix japonica*) exposed to single uranium doses as low as 11.9 mg/kg bw. Protein deposits in the kidneys, calcification of degenerating epithelial cells, calcified casts in the tubules, and damage to the collecting ducts were also observed. Mollenhauer *et al.* (1986) determined that the nephrotoxicity of uranium nitrate to birds is a result of its accumulation in the distal tubules and collecting ducts of the kidneys.

Sample *et al.* (1996) recommended an avian NOAEL of 16 mg/kg bw/d . This value was calculated by taking a dietary NOEC for black ducks of 1600 mg/kg food (Haseltine and Sileo 1983), multiplying by an average food consumption rate (125 g per day), dividing by the average weight of the ducks (1.25 kg), and applying an uncertainty factor of 10 (Sample *et al*. 1996).

There was a paucity of data on the effects of uranium in mammalian livestock or wildlife. Garner (1963) observed deterioration in the general health of cattle for 2 weeks with a concomitant decrease in milk yield, followed by a gradual return to an apparently normal state in spite of continued administration of 4 g uranium per day as uranyl nitrate. This value (4 g/d or 0.615 mg/kg bw/d, based on a weight of 650 kg [Puls 1994]) can be considered the lowest LOEL for domestic livestock.

A summary of all livestock and wildlife toxicity studies that were reviewed is included in Table 12. Toxicity studies on laboratory mammals (e.g., rats, mice, rabbits, guinea pigs, dogs), that may sometimes be used as surrogates for wildlife species, are described in Table 13.

CHAPTER 5. BEHAVIOUR AND EFFECTS IN HUMANS AND MAMMALIAN SPECIES

Although uranium can be found in three main isotopes in the Canadian environment: ^{234}U, ^{235}U and ^{238}U, the characteristics of the isotopes from a chemical toxicity perspective are felt to be similar. As a result, the discussion of the chemical effects of uranium in this chapter refers to "uranium" and covers all isotopes. More specifically, as discussed by Harley *et al.* (1999), the chemical characteristics of uranium (similar to other heavy metals) is independent of its isotopic form (this is in contrast to radiological properties where potencies do vary with isotopes). For the purposes of this evaluation, all isotopes of uranium were considered to demonstrate the same chemical and physical properties (e.g., chemical reactivity, melting point, boiling point and volatility) and consequently were considered to exhibit the same toxicological properties (i.e., were considered to be equipotent). This approach of treating various isotopes as equipotent is similar to the approach used for other heavy metals such as lead, cadmium and tungsten.

The behaviour and effects of uranium in humans and mammalian species has been reviewed by several major internationational health agencies including Health Canada (1999), WHO (2001a; 2004a), U.S. EPA (2005a; i.e., IRIS database) and ATSDR (1999). ATSDR (1999) provides the most detailed documentation on the toxicology of uranium whereas the Health Canada, WHO and U.S. EPA reports focus on the studies most relevant to the development of acceptable levels of exposure for regulatory purposes. This section of the report attempts to summarize some of the more relevant studies of uranium that were used in the development of the $PSQG_{HH}$.

Toxicokinetics

Absorption

Oral Exposure

Absorption of uranium via the oral route varies with the solubility of the uranium compound, the ingested dose (ATSDR 1999) and is also dependent on the animal species (Berlin and Rudell 1979).

Studies in animals show that the amount of uranium absorbed from the gastrointestinal tract is approximately 1% with the more soluble compounds (such as uranyl nitrate) absorbed more readily than the insoluble compounds (such as uranium oxides) (Sullivan 1980; Harrison and Stather 1981; Larsen *et al.* 1984; Wrenn *et al.* 1985; La Touche *et al.* 1987). In rats and rabbits, Tracy *et al.* (1992) reported that uranium in the form of uranyl nitrate hexahydrate was 0.06% absorbed across the gastrointestinal tract (when administered via drinking water).

In a review of the human oral absorption studies, ATSDR (1999) estimated that gastrointestinal absorption of uranium may range from <0.1 to 6% depending on the solubility of the uranium compound. Citing studies in human volunteers, ATSDR (1999) estimated that typical absorption rates of uranium was about 2% for uranium compounds found in food and water while ICRP provided estimates of 0.2% for insoluble uranium compounds and 2% for soluble uranium

compounds. In other studies, estimates of the uptake of uranium from the gastrointestinal tract of humans range from 0.01 to 30% (Hursh and Spoor 1973). Wrenn *et al.* (1985) reported that higher absorption rates tended to be associated with lower uranium intake levels. In human volunteers administered uranium in grapefruit drink, Karpas *et al.* (1998) reported oral absorption rates (0.1 to 1.6%) for a single dose of 100 µg. Adams and Spoor (1974) reported average absorption rates of 12% to 32% in the United Kingdom with intake levels of about 1 µg/d. Hursh *et al.* (1969) and Wrenn *et al.* (1985, 1988) estimated absorption rates of 0.5 to 8% at intake levels of 1.75 µg/d. Reviews by Harrison (1991), Leggett and Harrison (1995), and Wrenn *et al.* (1985) and a study by Spencer *et al.* (1990) show that the most likely oral absorption factor for uranium in humans is 1 to 2%. The estimates by Adams and Spoor (1974) were not included in the reviews on the basis of the uncertainties in the fluorometric measurements at low levels (Wrenn *et al.* 1985). In 50 human volunteers, Limson Zamora *et al.* (2002; 2003) estimated gastrointestinal absorption of uranium in humans at a median rate of 0.9% with a range of 0.1 to 6%. Finally, in a study involving 205 people living in Finland, Karpas *et al.* (2005) reported oral absorption factors for uranium from drinking water of 0.3% (median rate) with a range of 0.02 to 7%.

As is apparent from above, there is a wide range of estimates of oral absorption factors for uranium. Absorption seems to be dependent upon a number of factors including chemical form (e.g., soluble uranium compounds [such as uranium nitrate hexahydrate, uranium hexafluoride, uranyl fluoride, uranium tetrachloride, uranium pentachloride] seem to have greater bioavailability than insoluble forms [such as uranium oxides, uranium tetrafluoride]), the vehicle (medium) that it is presented in, and the animal species being evaluated. In addition, there has been suggestion that uranium found in soil might have lower bioaccessibility than uranium in the food and water studies discussed above (unpublished data from the Ontario MOE provided to the CCME). Overall, however, no specific studies on the possible reduced oral bioavailability of uranium and the various chemical species were identified in the published literature for development of $PSQG_{HH}$ and consequently a conservative approach was required in the estimation of the oral absorption estimates of uranium.

At this point in time, the level of precision on the possible reduced oral absorption rates for various chemical species of uranium and/or environmental media is not considered to be appropriate for development of the $PSQG_{HH}$ and instead an upper bound estimate of oral absorption was used. More specifically, the relative oral absorption factor for uranium in any chemical form in soil versus that for uranyl nitrate in the drinking water toxicology studies of Gilman *et al.* (1998 a,b) was considered to be 1 (i.e., uranium in soil regardless of the chemical species was considered to be equal to that in the toxicology studies). This is considered to be a conservative assumption that may overestimate the entry of uranium into the human body when used to estimate risks from contaminated sites (i.e., it has essentially been assumed that all uranium exists in a chemical form in soil that is as bioavailable as uranyl nitrate in drinking water). Although risk assessment professionals completing site-specific assessments may derive alternate values that consider the unique attributes of individual sites being evaluated, employment of such factors in the development of $PSQG_{HH}$ was not considered appropriate at the current time and instead a conservative approach was used that is intended to be protective of

all possible applications. If additional information becomes available in the future on the oral absorption of uranium, the $PSQG_{HH}$ could be revised to take into account such information.

Inhalation Exposure

In general, more soluble uranium compounds (such as uranyl fluoride and uranyl nitrate) are more likely to be absorbed rapidly following inhalation exposure than insoluble compounds (such as uranium oxides) (ATSDR 1999). In some cases, insoluble forms of uranium may remain in the lungs for years (ATSDR 1999). In addition to chemical form, the particle size of inhaled dusts of uranium will also affect the deposition and subsequent absorption of uranium across the respiratory tract (ATSDR 1999). Only particles of ≤ 10 µm AMAD (expressed as Activity Median Aerodynamic Diameter, AMAD) are deemed to be small enough to penetrate deep in the respiratory tract; larger particles (<10 µm AMAD) are expected to be cleared from the upper airways by mucocilliary action and subsequently swallowed into the GI tract (ATSDR 1999).

Absorption of uranium via the inhalation route is generally considered to be low and in the range reported for oral absorption. ATSDR (1999) summarized a number of studies that indicated that absorption of uranium across the respiratory tract may typically be in the range of 0.76 to 5% based on human studies in occupational settings but is highly dependent on the various physical attributes of the particulates (i.e., particle size, solubility). Based on data from Wrenn *et al.* (1985), absorption rates of inhaled uranium by mill workers were estimated to be 0.76% (range of 0.4 to 1.6%). ICRP (1996) estimated lung absorption factors of 2 to 4% in infants and 0.2 to 2% in adults.

Higher absorption rates across the respiratory tract have been noted in laboratory animal studies that typically involve administration of purified uranium compounds via aerosols (rather than as particulates that would be found at contaminated sites). In dogs, the estimated absorptions of uranium trioxide and uranium fluoride were found to be 23% and 42%, respectively, of the inhaled doses (Morrow *et al.* 1972; Morrow *et al*. 1982a). Rats exposed to uranyl nitrate aerosols (particle size ranging from 0.32 to 2.12 µm AMAD) by intracheal instillation retained up to 94% of the administered dose (Ballou *et al.* 1986), and in another experiment, rats absorbed at least 76% of the amount inhaled (Walinder *et al.* 1967). For uranium hexafluoride, Leach *et al.* (1984) reported absorption rates of 18 to 40% in rats and 20 to 31% in guinea pigs. Morrow *et al.* (1982b) also showed that the absorption of uranyl fluoride following intratracheal instillations (i.e., delivering uranium directly to the lower respiratory tract and bypassing the upper respiratory system) in dogs and rats was nearly complete.

Similar to that discussed for oral exposure, there is a wide range of estimates of inhalation absorption factors for uranium. Absorption seems to be dependent upon a number of factors that are difficult to account for in the development of $PSQG_{HH}$. For the purposes of development of the $PSQG_{HH}$, the relative inhalation absorption factor for uranium in any chemical form in suspended particulates was considered to be 1 (i.e., absorption of uranium in suspended particulates was considered to be equal to that in the oral toxicology studies). Although risk assessment professionals completing site-specific assessments may derive alternate values that consider the unique attributes of individual sites being evaluated, employment of such factors in the development of $PSQG_{HH}$ was not considered appropriate at the current time. If additional

information becomes available in the future on the inhalation absorption of uranium, the $PSQG_{HH}$ could be revised to take into account such information. It is noted that the inhalation route is a very minor pathway at most sites and appreciably greater (or lower) absorption values could have been used with little impact on the $PSQG_{HH}$.

Dermal Exposure

Little specific information has been identified that characterizes the absorption of uranium across the skin in humans and as a result most information is based on animal studies. Dunster (1964) estimated that 10% of a dermal dose of uranium penetrates the stratum corneum and enters the inner epidermis; however, details on this study were not available. Absorption of water-soluble uranium compounds by the skin has been shown to be about 0.1% (Wrenn *et al.* 1985). Uranyl nitrate, being both water and fat soluble, begins to penetrate the skin of experimental animals within 15 minutes of dermal application and is no longer found in the skin (epidermis or dermis) after 48 to 72 hours (de Rey *et al.* 1983). Orcutt (1949) and de Rey *et al.* (1983) observed toxic effects in experimental animals following dermal exposure to various uranium compounds, indicating some was absorbed.

For the purposes of development of the $PSQG_{HH}$, the relative dermal absorption factor for uranium in any chemical form in suspended particulates was considered to be 0.05 (i.e., absorption of uranium in suspended particulates was considered to be 5% of that in the oral toxicology studies). More specifically, if uranium is 0.1% absorbed across the dermal route but 2% across the oral route, the relative absorption factor would be 0.05 (i.e., 0.1/2 = 0.05). Although risk assessment professionals completing site-specific assessments may derive alternate values that consider the unique attributes of individual sites being evaluated, employment of such factors in the development of $PSQG_{HH}$ was not considered appropriate at the current time. If additional information becomes available in the future on the dermal absorption of uranium, the $PSQG_{HH}$ could be revised to take into account such information. It is noted that this was a relatively insensitive parameter in the development of the $PSQG_{HH}$ and appreciably greater (or lower) absorption values could have been used with little impact on the $PSQG_{HH}$.

Distribution

ICRP (1979) estimates that, following oral exposure and absorption of uranium from the gastrointestinal tract in humans, 10% is distributed to the kidneys, 10 to 20% is distributed to the bones, and the remainder is excreted in the urine. Dang *et al.* (1995) reported that the total concentrations in an urban Indian (Bombay) population living in a normal background environment were greatest in the skeleton, followed by muscle, soft tissue, lungs, kidney, liver, and then the heart. Given that the half-life of uranium is much greater in bones than other body parts, it is possible to have much greater concentrations in the skeleton than kidneys even though initial distribution rates were similar.

In experimental animals, once uranium has been absorbed following inhalation exposures, it is cleared from the blood very quickly for distribution to body tissues (Morrow *et al.* 1972; Eidson

et al. 1989; ATSDR 1999). The amount of uranium compounds absorbed via inhalation that are distributed to the skeleton has been reported to be 17 to 78% in rats (Walinder *et al.* 1967; Morrow *et al.* 1982b; Leach *et al.* 1984; Ballou *et al.* 1986), 34 to 43% in guinea pigs (Leach *et al.* 1984), and greater than 20% in dogs (Morrow *et al.* 1972). Uranium has been shown to accumulate in the tracheobronchial lymph nodes, lungs, bones, and kidneys of rats, dogs, and monkeys exposed to uranium compounds in the air (Walinder *et al.* 1967; Leach *et al.* 1973; Ballou *et al.* 1986). About 19 to 23% and 17 to 28% of inhaled uranium were distributed to the lung and kidneys of the rat, respectively, within one hour of exposure (Ballou *et al.* 1986). In rats, Tracy *et al.* (1992) reported that uranium in the form of uranyl nitrate hexahydrate (administered via drinking water) was distributed and retained in the skeleton and kidneys at rates comparable to parameters reported for humans.

No studies have been located regarding the distribution of uranium following dermal exposure in humans or animals. de Rey *et al.* (1984) reported that uranium dioxide implanted subcutaneously (thereby avoiding the skin barrier) freely penetrates the tissues and is promptly distributed to kidneys and bones via the vascular system. Once in the blood, uranium would be expected to be distributed similar to other routes of entry.

It is noted that ATSDR (1999) has reported that the solubility of uranium compounds may affect the distribution within the body where soluble uranium may primarly be deposited in the bone while insoluble uranium forms are deposited in the lungs and lymph nodes.

It is also noted that Chen *et al.* (2004) have found inhalation of uranium may result in a greater initial kidney burden than which may occur via the oral route. Using ICRP multicompartment biokinetic and dosimetric models, Chen *et al* (2004) generated a compilation of the kidney burdens following either acute inhalation or acute ingestion of the same amount of uranium and found greater initial concentrations in the human kidneys (particularly for soluble uranium compounds).

Half-life

Following inhalation exposure to beagle dogs, the half-life of soluble uranium compounds (ammonium diuranate) in the lungs was estimated to be 0.8 days in the first 4 days and estimated to be 30 days for periods thereafter (Eidson *et al.* 1989). [Note that the half-life here, and throughout the document, refers to pharmacokinetic clearance and not radiologic decay.] In contrast, the half-life of insoluble uranium compounds (U_3O_8) in the lungs of beagle dogs was estimated to be 2.7 days for 1% of the uranium and 2300 days for the remaining 99% (Eidson *et al.* 1989). Uranium trioxide had a half-life of 4.7 days in the lungs of dogs (Morrow *et al.* 1972), while uranyl fluoride had a half-life of 0.3 hours also in beagle dogs (Morrow *et al.* 1982b).

The half-life of uranium in the kidneys of humans has been estimated to be 1 to 6 days for 99% of the distributed dose and 1,500 days for the remainder (ICRP 1979). Dang *et al.* (1995) observed that the uranium burden in the kidney was 4 times higher than would be predicted by ICRP (1979), and suggested that the half-life should be longer. In rats, 95% of the dose distributed to the kidneys is excreted in the urine within one week (Sullivan 1980; Sullivan *et al.*

1986; La Touche *et al.* 1987). The half-life of uranium compounds in the kidneys has been reported to be 2 to 17 days in rats (Morrow *et al.* 1982b; Bentley *et al.* 1985) and 5 to 9 days in dogs (Morrow *et al.* 1982a; Eidson *et al.* 1989). Less than 1% of the absorbed amount remains in the kidneys after 30 days in dogs and rats (Ballou *et al.* 1986; Morrow *et al.* 1982a); after which time, the half-life for the remaining uranium in the kidneys of rats is approximately 100 days (Ballou *et al.* 1986).

The half-life of uranium in the bones is estimated to be 20 days for 99% of the distributed dose and 5,000 days for the remainder (ICRP 1979). Dang *et al.* (1995) support this estimate based on uranium burdens in bones of an urban Indian (Bombay) population living in a normal background environment. In rats, the biological half-life of uranium in the skeleton following inhalation exposure has been reported to be 150 to 200 days for uranyl nitrate (Ballou *et al.* 1986) and 69 days for uranyl fluoride (Morrow *et al.* 1982b). Bentley *et al.* (1985) reported a half-life of 21 days in the bones of the rat following parenteral injection of uranyl nitrate after 4 days; Eidson *et al.* (1989) reported a half-life of 50 days in bones of beagle dogs four days following inhalation exposure to ammonium diuranate.

It is noted that ATSDR (1999) has reported that the solubility of uranium compounds may affect the distribution within the body where soluble uranium may primarly be deposited in the bone while insoluble uranium forms are deposited in the lungs and lymph nodes.

It is also noted that Chen *et al.* (2004) have found inhalation of uranium may result in a greater initial kidney burden than which may occur via the oral route. In their analysis, Chen et al (2004) modeled the kidney burdens following either acute inhalation or acute ingestion of uranium and found greater initial concentrations in the kidneys of people (particularly for soluble uranium compounds).

Metabolism

According to ATSDR (1999), uranium can be metabolised to form different valencies (e.g., tetravalent uranium may be oxidized to hexavalent uranium and then uranyl ion) and then recomplexed (with citrates, bicarbonates or protein) to form other compounds. In the lung, small particles (< 4 nm) of uranium dioxide (UO_2) are oxidized to uranium trioxide (UO_3), and then form the uranyl ion by reacting with salt solutions (Cooper *et al.* 1982). Dissolved uranium is in the hexavalent state (UO_2^{2+}) in vivo, regardless of the oxidation state of the compound or the route of exposure (Hodge 1953; Eidson 1994). The uranyl ion enters the bloodstream and binds to components of the plasma: 50% to transferrin, 25% to citrate, and 25% to bicarbonate (Stevens *et al.* 1980; Cooper *et al.* 1982). The uranyl ion in plasma is in equilibrium with its complexes. The uranyl ion may also exist in the extracellular fluid (Hodge 1953). Uranium is transported to the tissues partly as a non-diffusable protein complex (about 40%) and partly as a diffusable bicarbonate complex (about 60%) (Hodge 1953). Uranium in the blood is removed primarily by the kidney (i.e., filetered from blood) and sequestered by the bones (Hodge 1953; Durbin 1960). In the glomerular filtrate, the low molecular weight complexes (bicarbonate and citrate) predominate. As bicarbonate, citrate, and water are reabsorbed in the proximal tubule, the concentration of uranyl ion in the tubular fluid increases. The uranyl ion then complexes

with phosphate ligands on the luminal surface of the tubule cells, resulting in enzyme inhibition, which suppresses cellular respiration and leads to slow cell death (BEIR IV 1988). In the bone, the uranyl ion competes with two calcium ions for position on the mineral surface (Neuman *et al.* 1949).

Elimination

Once in the blood, most uranium is excreted in the urine (i.e., two-thirds is excreted via the urine within 24 hours with an additional 10% within 5 days) while less than 1% is excreted in the faeces (ATSDR, 1999). On the other hand, most orally administered uranium is not absorbed and thus large amounts can be found in the faeces. Hursh *et al.* (1969), Spencer *et al.* (1990), and Wrenn *et al.* (1988) observed that most of the chronically ingested uranium at low levels in humans is excreted in the faeces, and the remainder (0.5 to 5%) is excreted in the urine. In rats, most ingested uranium (99%) remains non-absorbed and is eliminated in the faeces without being cycled through the bile (Sullivan 1980). Of the 1% that is absorbed, half is excreted in 7 days, and the other half is retained, primarily in the bone, for a longer period (Sullivan *et al.* 1986). Yu and Sherwood (1996) reported that workers at a gaseous diffusion plant eliminated uranium at relatively constant rates per unit air concentration (0.67 to 1.27 μg/L per $\mu g/m^3$).

Following inhalation in rats, about 60% of the absorbed amount of uranium is excreted in urine within 1 day (Morrow *et al.* 1982b; Leach *et al.* 1984; Ballou *et al.* 1986; Stradling *et al.* 1984, 1987). Morrow *et al.* (1982b) observed that 64% of inhaled uranyl fluoride in nose-exposed rats appeared in the faeces (probably as a result of translocation to the gut), while only 3 to 5% of the absorbed dose appeared in the faeces of similarly exposed dogs (probably as a result of biliary secretion of uranium). It is anticipated that over time, most of the remaining 40% will be excreted; however, some of the absorbed uranium will be retained, primarily in the lungs and bone (ATSDR 1999). Eidson *et al.* (1989) estimated that the urinary excretion rate for more soluble uranium compounds is higher than less soluble uranium compounds, but the faecal excretion rates are about the same for each.

No studies were located regarding the elimination of uranium following dermal exposure in humans or animals.

Mammalian Toxicology

The most sensitive indicator of uranium toxicity to mammals is renal toxicity. Table 13 provides a summary of toxicological data for some of the important studies completed on mammalian species.

It is noted in the review of the toxicology of uranium, the most important route of exposure in the development of the $PSQG_{HH}$ is the oral route. Although dermal and inhalation routes require consideration, the oral route represents the pathway of exposure which "drives" the development of the $PSQG_{HH}$. By deriving a $PSQG_{HH}$ that is protective of health effects via the oral route, effects from the dermal and inhalation route are also protected. This issue is addressed further in the discussion of uncertainties associated with the $PSQG_{HH}$ in Chapter 6.

Acute Toxicity

For many chemicals, acute toxicity is typically reported as LD_{50} (lethal dose, 50% mortality rate) or LC_{50} (lethal concentration, 50% mortality rate) values which represent the dose rate or concentrations that may cause a 50% mortality rate following a single exposure event. Such values typically are not tremendously useful in deriving $PSQG_{HH}$ since such endpoints are clearly unacceptable from a regulatory perspective. In addition, health protection to prevent chronic toxicity is typically conservative enough to protect against acute toxic endpoints (unless there is unusually high episodic exposures). Nevertheless, some of the most important acute toxicity information is summarized below to provide context into the health effects that may be encountered following single dose exposures to large amounts of uranium.

Oral Studies

The lowest LD_{50} values (i.e., dose associated with a 50% mortality rate) reported were 114 and 136 mg/kg bw/day for rats and mice, respectively, that were administered a single dose of uranyl acetate dihydrate by gavage (Domingo *et al.* 1987). Rats exhibited minimal hepatic lesions, renal proteinura, and body weight loss following a single oral dose of 130 mg/kg bw/d (Domingo *et al.* 1987). Physical signs observed in rats following a single oral dose of uranium included tremors, piloerection, and a decrease in pupillary size (Domingo *et al.* 1987).

Inhalation Studies

The lowest LC_{50} values reported for inhalation exposure to uranium (as uranium hexafluoride) were 12 000 mg/m^3 in rats for a 10-minute duration and 62 000 mg/m^3 in guinea pigs for a 2-minute duration (Leach *et al.* 1984).

Renal effects such as increased lactate dehydrogenase (LDH), polyuria, glucosuria, and proteinuria were observed in rats and guinea pigs exposed to uranium in the air (Leach *et al.* 1984). The lowest No-Observable-Effect Concentration (NOEC) and Lowest-Observable-Effect-Concentration (LOEC) (glucosuria and increased LDH) values reported were 1,360 mg/m^3 in rats exposed for 2 minutes and 580 mg/m^3 in rats exposed for 5 minutes, respectively (Leach *et al.* 1984).

Dermal Studies

Animals dermally exposed to uranium for 4 hours had the following LD_{50} values (based on applied doses): 28 mg/kg bw for rabbits; 1190 mg/kg bw for guinea pigs; and 4286 mg/kg bw for mice (Orcutt 1949). Renal effects such as renal failure, moderate injury, and proteinuria were observed following acute dermal exposure to uranium (Orcutt 1949; de Rey *et al.* 1983). Weight loss has also been observed in mice, guinea pigs, rabbits and rats following acute dermal exposure (Orcutt 1949). Mild lesions and moderate skin irritation have been observed following acute dermal exposure to uranium in rats and rabbits. Lopez *et al.* (2000) reported lethal effects of uranium administered topically to rats (as uranyl nitrate) were dependent upon dose with both area of skin exposed and duration of exposure affecting the lethality of uranium.

Sub-chronic Toxicity

Some of the more important studies relevant to the sub-chronic toxicity of uranium are discussed below; however, in all cases, a weight of evidence approach is recommended in interpretation of toxicological data. ATSDR (1999) provides a detailed review of the various toxicological data on sub-chronic toxicity of uranium.

Oral Studies

The lowest 30-d LD_{50} value for uranium was 386 mg/kg bw/d for rats (Maynard and Hodge 1949). Renal effects including microscopic lesions, congestion, moderate damage, and necrosis were observed in various studies following short-term oral exposure to uranium (Maynard and Hodge 1949; Tannenbaum and Silverstone 1951; Goel *et al.* 1980; Ortega *et al.* 1989; Gilman *et al.* 1998a,b). Hepatic effects including microscopic lesions, congestion, and cloudy swelling were observed in various short-term studies (Tannenbaum and Silverstone 1951; Goel *et al.* 1979; Ortega *et al.* 1989). Hematological effects including increased hematocrit were observed in rats and mice (Ortega *et al.* 1989). Other effects of short-term oral exposure to uranium include decreased body weight gain or weight loss in rats, rabbits and mice (Maynard and Hodge 1949; Tannenbaum and Silverstone 1951).

The lowest NOEL value reported for short-term renal effects was 4 mg/kg bw/d in rats orally exposed to uranium for 30 days (Maynard and Hodge 1949). Gilman *et al.* (1998a) reported LOEL values of 0.05 and 0.49 mg/kg bw/d in male and female rabbits, respectively, exhibiting changes in tubular nuclei when exposed to contaminated drinking water for 91 days. Several of the male rabbits in this study developed bacterial infections during the experimental period, and therefore the LOEL value for male rabbits should be considered with caution. In a similar 91-day rat study, Gilman *et al.* (1998c) reported a LOAEL for degenerative lesions in the proximal convoluted tubule of the kidney in male rats to be 0.06 mg/kg bw/d following exposure to uranyl nitrate hexahydrate in drinking water. For female rats exposed to uranium through drinking water, a 91-d LOAEL of 0.09 mg/kg bw/d was reported for degenerative lesions in the glomerulus of the kidney (Gilman *et al.* 1998c).

Inhalation Studies

Inhalation exposures to uranium may result in renal, ocular, hematological and pulmonary effects in animals (ATSDR 1999). In general, rabbits and cats have been identified as the most sensitive species to uranium via inhalation. Serious effects have been reported in animals exposed to uranium in the range of 2 to 18 mg U/m^3 with soluble uranium. Insoluble uranium may also result in lethality but at higher concentrations than reported for soluble uranium compounds (ATSDR 1999). Sub-chronic exposures (6 hr/d for 30 days) to uranium hexafluoride at concentrations of 2 mg U/m^3 resulted in mortality in guinea pigs (5%), dogs (20%) and rabbits (80%) (Spiegl 1949). Sub-chronic exposures (8 hr/d, 5 d/wk for 30 days) to uranium hexafluoride at concentrations of 2 mg U/m^3 resulted in 100% mortality in cats (Roberts 1949).

ATSDR (1999) has reported NOAEC in the range of 0.2 to 19 mg U/m^3 depending upon the animals evaluated and the chemical form of uranium. Respiratory, ocular and renal effects appear to be the most sensitive indicators of sub-chronic effects from inhalation studies. Following inhalation exposures, renal effects such as mild and moderate tubular injury, and mild tubular atrophy following short-term inhalation exposure have been observed in a variety of animals. Other short-term effects resulting from inhalation exposure to uranium include haematological effects and decreased body weight gain in rabbits.

Dermal Studies

Death has been reported to occur in guinea pigs dermally exposed to uranium for 4 weeks at an applied dose of 379 mg/kg bw/d (Orcutt 1949). Other effects including renal proteinuria, weight loss, severe dermal ulcers, and skin irritation were reported in guinea pigs and rabbits (Orcutt 1949). The lowest LOEC reported was 2.3 mg/kg bw/d for proteinuria, transitory weight loss, and severe dermal ulcers in rabbits exposed for 5 weeks (Orcutt 1949).

Long-term/Chronic Toxicity

Some of the more important studies relevant to the chronic toxicity of uranium are discussed below; however, in all cases, a weight of evidence approach is recommended in interpretation of toxicological data. ATSDR (1999) provides a detailed review of the various toxicological data on sub-chonic toxicity of uranium.

Oral Studies

Lifetime feeding studies with uranium (as uranyl fluoride, uranyl nitrate hexahydrate and uranium dioxide) have indicated that exposure to high amounts of uranium can lead to decreased longevity of laboratory animals (Maynard and Hodge 1949). A NOEL (for longevity) was reported to be 81 mg U/kg bw/d for rats exposed to uranyl fluoride. For other uranium compounds, exposures of 1,130 mg U/kg bw/d (as uranyl nitrate), 1,390 mg U/kg bw/d (as uranium tetrafluoride) and 1,630 mg U/kg bw/d (as uranium dioxide) were reported as NOELs for decreased longevitiy (Maynard and Hodge 1949; Maynard *et al.* 1953). Most deaths were associated with chemically-induced renal damage. Renal effects including tubular necrosis, tubular lesion, and mild tubular degeneration resulting from chronic oral exposure to uranium in

rats have been observed (Maynard and Hodge 1949; Maynard *et al.* 1953). Hematological effects included anemia, and increased white blood cell count (Maynard and Hodge 1949; Maynard *et al.* 1953). Other chronic effects resulting from oral exposure to uranium include a decrease in body weight gain in rats. No histological effects were found in the spleen, lymph nodes, or bone marrow of various animals orally exposed (Maynard and Hodge 1949; Tannenbaum and Silverstone 1951; Maynard *et al.* 1953). The lowest NOEL value reported is 24 mg/kg bw/d for chronic renal effects in rats orally exposed to uranyl nitrate hexahydrate for one year (Maynard *et al.* 1953). The lowest reported NOEL and LOEL values for haematological and renal effects were 19 and 39 mg/kg bw/d, respectively, in rats exposed to uranyl nitrate hexahydrate for two years (Maynard *et al.* 1953).

Inhalation Studies

Leach *et al.* (1970, 1973) exposed dogs and monkeys to natural uranium dust (as uranium dioxide aerosols of approximately 1 μm mass median particle diameter) for 5 days per week, 5.4 hours per day, for 1 to 5 years. At 5.1 mg U/m^3 (corresponding to 5.8 mg UO_2/m^3) both species exhibited lymph node fibrosis. Stokinger *et al.* (1953) did not observe any significant histological changes in the lymph nodes, bone marrow, or spleen of rats, rabbits, guinea pigs, or dogs exposed to various forms of soluble and insoluble uranium at concentrations in the range of 0.15 to 2 mg/m^3; nor was there a significant build-up of uranium in these tissues. No deaths were observed in rats and dogs chronically exposed to uranium concentrations as high as 10 mg/m^3 (Stokinger *et al.* 1953). Renal effects such as mild tubular injury in rats and dogs were observed following chronic inhalation exposure to uranium (Stokinger *et al.* 1953).

No hematological effects were observed in rats and dogs exposed to uranium concentrations as high as 10 mg/m^3 for a 1-year duration (Stokinger *et al.* 1953) and 5.1 mg U/m^3 in monkeys for 1 to 5-year durations (Leach *et al.* 1970, 1973). The concentration of 0.05 mg/m^3 was both the lowest NOEC value determined in guinea pigs and the lowest LOEC determined in rats for mild tubular injury following exposure to uranium hexafluoride in the air for one year (Stokinger *et al.* 1953).

Dermal Studies

No studies were located regarding chronic effects in animals following dermal exposure to uranium.

Carcinogenicity

In discussing the possible carcinogenicity of uranium, it is important to distinguish the chemical effects from those due to possible radiation. As discussed earlier, this document only addresses health issues associated with the chemical toxicity of uranium. With respect to chemical toxicity, there is insufficient evidence to conclude that oral, inhalation, or dermal exposure of natural uranium will cause cancer in animals (ATSDR 1999; U.S. EPA 1998).

Leach *et al* (1970, 1973) found that dogs exhibited lung neoplasms and atypical epithelial proliferation when exposed to uranium dioxide in the air at 5. 1 mg U/m^3 for 5 d/wk, 5.4 h/d, for 1 to 5 years. The authors recommended against extrapolating the findings to humans since the glandular neoplasms do not appear to be particularly common in humans.

Genotoxicity

There is a paucity of information on the mutagenic effects in microbial systems or in animals following oral, inhalation, or dermal exposure to uranium. To assess the potential mutagenic effects of long-term exposure to internalized depleted uranium,

Miller *et al.* (1998) implanted depleted uranium into rats and then evaluated urine and serum for mutagenic potential at various times after pellet implantation using the Ames Salmonella reversion assay. Enhancement of mutagenic activity was observed in urine samples from animals implanted with depleted uranium pellets while sera samples did not indicate any evidence of mutagenicity.

Lin *et al.* (1993) examined the cytogenetic effect of uranium in Chinese Hamster Ovary (CHO) cells. The authors found uranyl nitrate at concentrations ranging from 0.01 to 0.3 mM may result in decreased cell cycle kinetics and increased frequencies of micronuclei and chromosome aberrations. The authors concluded that uranyl nitrate has the ability to cause genotoxicity and cytotoxicity in CHO cells.

Reproduction and Teratology

Oral Studies

Although a true NOEL has not been identified for oral exposures to uranium causing developmental effects, it appears that such effects are less sensitive than other effects that have been associated with uranium (ATSDR 1999).

Domingo *et al.* (1989a) exposed pregnant mice to uranium (as uranyl acetate dihydrate) in water at doses of 0.028, 0.28, 2.8 and 28 mg U/kg bw/d. Uranium exposure of 28 mg U/kg bw/d to pregnant mice during gestation days 6 to 15 resulted in decreased maternal body weights, stunted fetuses and skeletal malformations of fetuses (Domingo *et al.* 1989a). In a separate study, exposure of 3 mg/kg bw/d to female mice from gestation day 13 to lactation day 21 resulted in maternal death (Domingo *et al.* 1989b). Female rats exposed to a single dose of 474 mg/kg bw/d had fewer pups than controls (Maynard *et al.* 1953).

Short-term exposures to uranium resulted in testicular lesions in male rats at 66 mg/kg bw/d for a 4-month duration (Malenchenko *et al.* 1978). An increase in resorption in mice exposed before, during, and after pregnancy for a duration of 4 to 8 weeks was observed at 14 mg/kg bw/d, as well as decreases in pup weight and growth and pup mortality (Paternain *et al.* 1989). Chronic exposures to uranium in male rats resulted in changes in testes histopathology at 474 mg/kg bw/d after one year and testicular atrophy at 97 mg/kg bw/d after two years (Maynard and Hodge 1949; Maynard *et al.* 1953). Females that mated with male rats fed uranyl acetate dihydrate in

drinking water for 64 days exhibited a decrease in pregnancy rate at 10 mg/kg bw/d, but had no adverse effect on testicular function (Llobet *et al.* 1991).

Inhalation Studies

No significant histological changes were found in the ovaries of rats exposed to uranium dioxide in the air for periods ranging from 1 to 5 years (Leach *et al.* 1970, 1973).

Dermal Studies

No studies were located regarding reproductive or teratogenic effects in animals following dermal exposure to uranium.

Human Toxicology

Acute Toxicity

Limited data are available on the human health effects resulting from acute oral, inhalation, or dermal exposure to uranium. Single intravenous doses of hexavalent uranium of 120 µg U/kg bw and higher administered to terminal brain tumour patients were associated with elevations in urinary excretion of catalase, albumin and non-protein nitrogen, and casts in the urine (Luessenhop *et al.* 1958). Trace changes in urinary catalase were observed in patients injected with 55 µg/kg bw as uranyl nitrate (Hursh and Spoor 1973). In workers accidentally exposed to a high concentration of airborne uranium hexafluoride and its hydrolysis products (including the highly toxic hydrofluoric acid), survivors experienced injuries to the eyes, respiratory tract, skin, and gastrointestinal tract (Kathren and Moore 1986). In another accident, a worker briefly exposed to uranium tetrafluoride through inhalation later exhibited symptoms of kidney damage (Zhao and Zhao 1990).

Sub-chronic Toxicity

The human health effects resulting from sub-chronic oral, inhalation, or dermal exposure to uranium are not known (ATSDR 1999).

Long-term/Chronic Toxicity

In Nova Scotia, residents who drank water containing up to 0.7 mg/L uranium from private wells exhibited no overt renal disease attributed to the exposure (Moss *et al.* 1983; Moss 1985). A dose-response effect of increased β2-microglobulin excretion was observed with increasing uranium exposure, with evidence of reversibility for individuals who had stopped using the uranium-contaminated drinking water (Moss *et al.* 1983; Moss 1985). These studies were subsequently updated in an article by Zamora *et al.* (1998) where intakes in the range of 0.004 to 9 µg/kg bw/d (via drinking water) were postulated to affect kidney function and that the proximal tubule, rather than the glomerulus, was the site for this interference. Study limitations have prevented the use of these data in the development of a TDI for uranium by any major health agency reviewed.

Mao *et al.* (1995) evaluated the possible correlation between uranium exposure and microalbuminuria. In their analysis, Mao *et al.* (1995) investigated the association between uranium concentrations in drinking water and microalbuminuria (a sensitive biological indicator of renal dysfunction). The evaluation involved 100 participants from 3 communities in Saskatchewan. Mao *et al.* (1995) found a positive association between uranium exposure and urine albumin levels. It is noted that this was not necessarily an indicator of renal toxicity since urine albumin levels were still within the range of normal (i.e., the degree of microalbuminuria did not appear to be clinically significant). In addition, it is noted some of the persons in the communities evaluated were exposed to drinking water concentrations of 20 µg uranium/L (as a mean value) and up to 50 µg/L (as a maximum value) which is greater than the current Canadian Drinking Water Guideline of 20 µg/L. Russell *et al.* (1996) examined histopathological slides of kidneys obtained at autopsy from 7 uranium workers known to have been exposed to low levels (few mg to few hundred mg) of the metal over several years to be compared with similar slides from 6 reference cases without uranium exposure. The nephrotoxic effects of uranium on the kidney were not observed in any of the tissues.

Several studies reported an increase in deaths resulting from cancers of lymphatic and hematopoietic tissue in uranium millers and workers in a uranium enrichment facility (Archer *et al.* 1973b; Brown and Bloom 1987). Other studies of workers in uranium enrichment plants or weapons facilities, however, reported no increased incidence of death due to cancers of the immune system (Polednak and Frome 1981; Acquavella *et al.* 1985; Checkoway and Crawford-Brown 1987).
Several studies were located that investigated sex ratios in the offspring of uranium miners (Müller and Ruzicka-Jaroslav Bakstein 1967; Waxweiler *et al.* 1981b; Wiese and Skipper 1986). No significant effects due to non-radiological exposures on the sex ratio were found in any of the studies, but one study reported an increase in first-born female offspring in the miners (Müller and Ruzicka-Jaroslav Bakstein 1967). No effects on frequency of low birth weight of infants, miscarriages, or fertility were reported in uranium miners (Wiese and Skipper 1986).

Retrospective epidemiological studies of male workers at uranium mill and metal processing plants showed no increase in overall deaths due to exposure to uranium, and a lower incidence of some causes of death were reported (due to "healthy worker" effect) (Scott *et al.* 1972; Archer *et*

al. 1973b; Polednak and Frome 1981; Hadjimichael *et al.* 1983; Waxweiler *et al.* 1983; Acquavella *et al.* 1985; Brown and Bloom 1987; Checkoway and Crawford-Brown 1987; Cragle *et al.* 1988). Mortality from respiratory disease was greater in workers in uranium mill, fabrication, processing and enrichment plants than in controls (Archer *et al.* 1976; Polednak and Frome 1981; Hadjimichael *et al.* 1983; Waxweiler *et al.* 1983; Dupree *et al.* 1987). Respiratory diseases included obstructive pulmonary disease (Hadjimichael *et al.* 1983; Waxweiler *et al.* 1983), emphysema, fibrosis, and silicoses (Archer *et al.* 1976; Waxweiler *et al.* 1983). Mortality from renal disease was greater in uranium millers and miners than in controls (Waxweiler *et al.* 1981a, 1983). An increase in deaths from lung cancer was found in workers in a uranium enrichment plant who were not exposed to radon gas (thereby eliminating radiological effects of radon and its decay products) (Polednak and Frome 1981).

Carcinogenicity

Uranium has been classified as a Group VA carcinogen (inadequate data for evaluation) by Health Canada (1996). U.S. EPA's carcinogenicity assessment has been withdrawn pending further review (U.S. EPA 2005a). ATSDR does not currently assess cancer potency or perform cancer risk assessments. Although (WHO 2004a) seems to suggest that cancer following ingestion of uranium has not been shown, no formal WHO cancer assessment has been provided. It is noted that certain occupational groups consider uranium to be a confirmed carcinogen (ACGIH 1998) or potential carcinogen (NIOSH 1992); however, these assessments may also include the radiological issues associated with uranium. Overall, none of the major international health agencies reviewed (i.e., Health Canada, U.S. EPA, WHO or ATSDR) currently classify uranium as a carcinogen and no cancer slope factors are currently provided by these agencies with regard to the chemical properties of uranium. Nevertheless, the positions of these agencies are suggestive that the information should be regularly reviewed as new data are collected.

Oral Exposure

No studies were located regarding carcinogenicity in humans following oral exposure to uranium.

Inhalation Exposure

Retrospective epidemiological studies of uranium miners have found an increase in mortality due to lung cancer (Lundin *et al.* 1969; Archer *et al.* 1973a; Grace *et al.* 1980; Gottlieb and Husen 1982; Samet *et al.* 1984; Saccomanno *et al.* 1986; Howe *et al.* 1986, 1987; Gilliland *et al.* 2000); however, the uranium miners in all of these studies were also exposed to other suspected carcinogens, such as radon. Only one study reported an increase in deaths from lung cancer in workers who were not exposed to radon (Polednak and Frome 1981). It is difficult to determine if inhalation exposure to natural uranium can cause cancer in humans since most epidemiological studies involve multiple chemical exposures (ATSDR 1999).

Dermal Exposure

No studies were located regarding carcinogenicity in humans following dermal exposure to uranium.

Overall Toxicological Evaluation

According to Health Canada (1994; 1996), the term Tolerable Daily Intake (TDI) refers to the intake of a chemical to which it is believed that a person can be exposed daily over a lifetime without deleterious effect." In other words, the TDI is the amount of exposure that is considered to be unlikely to cause adverse health effects in the general population, including sensitive individuals, but excluding those with allergy or other hypersensitivity. The TDI is, effectively, the best estimate of the human threshold dose, considering uncertainties and variability in intra-species (inter-individual) toxic response, inter-species toxic response (where toxicity data from animal studies is extrapolated to the human population) and limitations of the toxicological database. TDIs are usually provided as daily dose rates in units of mass of chemical per kilogram of body weight of a person per day (e.g., mg/kg body weight (bw)/day). Other terms that are analogous to the Tolerable Daily Intake include Tolerable Intake (TI; used by the World Health Organization [WHO]), Reference Dose (RfD; used by the U.S. Environmental Protection Agency [U.S. EPA]) and Minimum Risk Level (MRL; used by the Agency for Toxic Substances and Disease Registry [ATSDR]).

In the case of uranium, Health Canada, ATSDR, U.S. EPA and WHO all offered toxicity reference values (TRVs) that were considered in the development of the $PSQG_{HH}$ and these are discussed below. Health Canada (1999) derived a TDI for uranium of 0.6 µg/kg bw/day based on a subchronic study using rats. Several toxicological studies have been completed in various species of laboratory animals (mice, rats, rabbits and dogs) exposed to uranium that range in duration from less than one month of exposure to up to 2 years. According to Health Canada (1999), rats are the most sensitive species to uranium and a rat study by Gilman *et al.* (1998a) represents the most appropriate study for estimation of the TDI (i.e., this is the most appropriate assay that provides the lowest TDI). In Gilman *et al.* (1998a), male and female rats were exposed to uranium in drinking water in the form of uranyl nitrate hexahydrate. The Lowest Observed Adverse Effect Level (LOAEL) for degenerative lesions in the proximal convoluted tubule of the kidney in male rats was found to be 60 µg/kg bw/day as uranium (Gilman *et al.*, 1998a). In this study, female rats were slightly less responsive to uranium than male rats (LOAEL of 90 µg/kg bw/day) which is a sex-related pattern that has also been observed in rabbits (Gilman *et al.*, 1998b). An uncertainty factor of 100 was then applied to the LOAEL of 60 µg/kg bw/day to take into account sensitive populations (10-fold for intra-species variation) and extrapolation from animals to human studies (10-fold for interspecies variation). Health Canada (1999) indicated that an additional uncertainty factor for use of a LOAEL rather than a NOAEL was not necessary due to minimal severity of the lesions reported. In addition, Health Canada (1999) indicated that the use of a subchronic study for estimation of a TDI was adequately sensitive and did not require an additional uncertainty factor. As a result, Health Canada (1999) estimated a TDI of 0.6 µg/kg bw/day as maximum acceptable exposure for protection of the general population.

It is noted that Health Canada (1999) did not use the rabbit study of Gilman *et al.* (1998b) to

estimate their TDI even though this study had a slightly lower LOAEL. Gilman *et al.* (1998b) reported a LOAEL of 50 µg/kg bw/day for renal toxicity for male rabbits exposed to uranium in drinking water (as uranyl nitrate hexahydrate) (female rabbits were less sensitive to uranium with a LOAEL of 490 µg/kg bw/day). Although the rabbit study was associated with a more conservative LOAEL, Health Canada (1999) did not consider this study to be as reliable as Gilman *et al.* (1999) due to *Pasturella multocida* infection in male rabbits potentially confounding the results. Nevertheless, it is noted that the differences in the reported LOAELs were not great between rats and rabbits.

It is also noted that Health Canada (1999), WHO (2001a; 2004a), U.S. EPA (2005a) and ATSDR (1999) have all reported that the use of less than lifetime exposures do not require additional uncertainty factors for estimation of toxicity reference. These health agencies have concluded that subchronic exposures to uranium are considered to be adequately sensitive for determining doses that cause chronic renal toxic effects. It seems to be consensus that the toxicity of uranium seems to be more dependent on the uranium dose administered rather than the duration of exposure.

In the case of the ATSDR, a MRL of 2 µg/kg bw/day was estimated based on the rabbit study by Gilman *et al.* (1998b). ATSDR (1999) applied an uncertainty factor of 30 to the LOAEL of 50 µg/kg bw/day to account for data deficiencies (3-fold use of a minimal LOAEL instead of a NOAEL) and sensitive populations (10-fold for intra-species variation). ATSDR (1999) noted that an additional uncertainty factor for extrapolation from animals to human studies was not required since they considered rabbits to be more sensitive than people. Based on this rationale, ATSDR (1999) estimated a MRL of 2 µg/kg bw/day for protection of the general population. Although developed for intermediate exposure durations (i.e., exposures up to one year in duration), the ATSDR noted that this MRL should also be considered to be protective of long term (i.e., lifetime) exposures (see discussion below for more details).

U.S. EPA (2005a) currently recommends a RfD of 3 µg/kg bw/day which is the least conservative toxicity reference value of all major international health agencies that have been reviewed. U.S. EPA estimated a RfD based on the study by Maynard and Hodge (1949) whereby rabbits were administered uranyl nitrate hexahydrate in food for 30 days. In this study, a LOAEL of 2,800 µg/kg bw/day (effects were initial body weight loss and moderate renal toxicity). U.S. EPA then reported an uncertainty factor of 1,000 to the LOAEL of 2,800 µg/kg bw/day to take into account data deficiencies (10-fold use of a minimal LOAEL instead of a NOAEL), sensitive populations (10-fold for intraspecies variation) and extrapolation from animals to human studies (10-fold for interspecies variation). It is noted the U.S. EPA (2005a) reports that their RfD has not been revised since 1989 and it is not entirely clear if the U.S. EPA has considered the more recent toxicological studies of Gilman *et al.* (1998 a, b) that have been used by Health Canada, WHO and the ATSDR to develop their toxicity reference values.

Finally, the WHO currently has provided 2 different estimates of tolerable daily exposures for uranium. WHO (2001a) reported a Tolerable Intake (TI) of 0.5 µg/kg bw/day for depleted uranium. This TI was estimated based on the rabbit study by Gilman *et al.* (1998b) (i.e., same study as used by ATSDR [1999]) whereby a LOAEL of 50 µg/kg bw/day for renal toxicity was

estimated in male rabbits. WHO (2001a) applied an uncertainty factor of 100 to the LOAEL of 50 µg/kg bw/day to take into account data deficiencies (3-fold use of a minimal LOAEL instead of a NOAEL), sensitive populations (10-fold for intraspecies variation) and extrapolation from animals to human studies (3-fold for interspecies variation that account for toxicodynamic and toxicokinetic differences).

On the other hand, WHO (2004a) provides a TDI of 0.6 µg/kg bw/day for uranium in drinking water that is based on an identical rationale as provided by Health Canada (1999).

Overall, the Health Canada TDI for uranium was considered to be appropriate for use in the development of a $PSQG_{HH}$. In the cases of the ATSDR (1999) and U.S. EPA (2005a), these international health agencies developed toxicity reference values that were considerably less conservative than the Health Canada value while WHO (2001a) offered a value that was only slightly more conservative, and likely insignificantly so given the uncertainties in the determination of toxicological reference values. Finally, WHO (2004a) offers a TDI value that is identical to Health Canada (1999).

Based on the above, the Health Canada TDI of 0.6 µg/kg bw/day was considered to be appropriate for the purposes of establishing soil quality guidelines.

Summary of Various Toxicity Reference Values (TRVs) for Uranium

Health Agency	Animal Species Used to Estimate TRV	Assumed LOAEL (µg/kg bw/day)	Endpoint	Uncertainty Factor Used to Estimate TRV	Estimated TRV (µg/kg bw/day)
Health Canada (1999)	Rat	60	Renal toxicity	100	0.6
ATSDR (1999)	Rabbit	50	Renal toxicity	30	2
U.S. EPA (2005a)	Rabbit	2,800	Decreased body weight, renal toxicity	1,000	3
WHO (2001a)	Rabbit	50	Renal toxicity	100	0.5
WHO (2004a)	Rat	60	Renal toxicity	100	0.6

In the case of inhalation exposures, ATSDR (1999) and WHO (2001a) are the only major international health agencies that have toxicity reference values for this pathway. ATSDR (1999) recommends a MRL of 0.0003 mg U/m^3 (or 0.3 µg U/m^3) for protection of chronic continuous exposures to soluble uranium and 0.008 mg U/m^3 (or 8 µg U/m^3) for protection of intermediate exposures to insoluble uranium. The ATSDR MRL of 0.3 µg U/m^3 for soluble uranium was based on a NOAEC of 0.05 mg U/m^3 in dogs in the uranium tetrachloride study by Stokinger *et al.* (1953) (where minimal microscopic lesions in the renal tubules were observed at a LOAEC of 0.20 mg U/m^3) and then application of a 30-fold uncertainty factor to account for interspecies and intraspecies variability and additional factors to account for continuous exposures and differences between dog and human respiratory dynamics. The ATSDR MRL of 8 µg U/m^3 for insoluble uranium was based on a NOAEC of 1.1 mg U/m^3 in dogs in the uranium dioxide study of Rothstein *et al.* (1949) (where minimal microscopic lesions in the renal tubules were observed at a LOAEC of 8.2 mg U/m^3) and then application of a 30-fold uncertainty factor to account for

interspecies and intraspecies variability and additional factors to account for continuous exposures and differences between dog and human respiratory dynamics.

In addition to the ATSDR, WHO (2001a) stated that long term exposure to concentrations less than 10 mg/m^3 have usually not resulted in pulmonary toxicity. WHO (2001a) also concludes that toxicological data for the rat indicates a time adjusted NOAEL of 20 µg U/m^3 which is equivalent to a dose rate of 60 µg/kg/day.

Due to a lack of more specific health agency guidance, the Health Canada TDI for the oral route was also used to address the inhalation route as this would mathematically be more conservative than addressing the inhalation route separately (especially since both oral and inhalation exposures have renal effects as the most sensitive indicator of toxicity). It is noted that the use of a TDI that is not specific to the inhalation route was not a major contributor to uncertainty in the overall guideline development process (i.e., dust concentrations associated with $PSQG_{HH}$ were much lower than air concentrations considered acceptable by ATSDR and WHO and thus did not appreciably affect the derived values).

Human Exposure Estimates

The Estimated Daily Intake (EDI) is expressed in units of "µg/kg bw/day" and is intended to represent the average exposure that a Canadian may receive to uranium. The normal sources from which a person may receive exposure to uranium is primarily thought to include foods, soils, air and water. No specific consumer products were identified as a source of additional background uranium exposure in the general Canadian population. Employing average concentrations of uranium in the various media, and the typical rates of intake of those media for the Canadian population, the EDI for uranium was derived.

To estimate the EDI for uranium, the general population was subdivided into four age classes including preschoolers (7 months to 4 years), school age children (5 to 11 years), teenagers (12 to 19 years), and adults (20 years and older). Infant exposure (0 to 6 months) was not assessed as there was no data available for uranium concentrations in breast milk or formula. Table 14 shows reference body weights and standard intakes of air, drinking water and soil (including dust) for each specified age class of the population. Canadian food consumption rates are summarised in Table 15 for 11 food composite groups. Uranium intake through dietary consumption was estimated using food concentration data from the 2001 UK Total Diet Study (Table 7). Total daily intakes of uranium by all routes of exposure were calculated by age class and are presented in Table 17.

The typical uranium concentration used in the exposure estimates for air was 0.0001 µg/m^3. This value represents background atmospheric uranium concentrations in southern Ontario (Tracy and Prantl 1985), the only Canadian data identified. This air concentration is at the upper limit of the background uranium concentration range of 0.000025 to 0.0001 µg/m^3 reported by NCRP (1999) and higher than the mean concentration of 0.000076 µg/m^3 reported in 1985/1986) for New York City (Fisenne *et al.* 1987). Since no data on uranium concentrations in indoor air were identified, ambient air concentrations were used as best estimates of concentrations in indoor air.

For drinking water consumption, an urban exposure scenario is the most common situation expected to arise as 77% of Canadians live in cities (Statistics Canada 1993) and 82% of these urban dwellers receive treated water supplies, mostly from surface water sources (Tate and Lacelle, 1992). A concentration of 0.2 µg/L was used as the typical uranium level in Canadian drinking water, based on a survey of uranium in drinking water (n=258) collected between 1998 to 2002 from 129 lake and river water treatment plants under the Ontario Drinking Water Surveillance Program (P. Cheung, Ontario MOE, pers. comm.). Slightly higher uranium concentrations were reported for drinking water from groundwater sources in Quebec (geometric means ranging from 0.35 to 0.97 µg/L) (Choinière and Beaumier, 1997), and the Yukon (up to 7.2 µg/L) (E. Bergsam, Yukon Department of Environmental Health, pers. comm.). However, the Quebec water samples are dated (sampled between 1974 and 1982) and the Yukon data set contained relatively few samples (n=18). In addition, both data sets were limited by high detection limits.

A background soil concentration of uranium of 2 mg/kg was assumed, both for purpose of deriving estimated daily (background) intake from soil, and as the starting point for the derivation of the hypothetical soil quality guidelines derived herein. This concentration is reflective of the uranium concentration measured in background soils collected in Ontario and New Brunswick (Gordon, 1992; OMEE, 1993; Pilgrim and Schroeder, 1997; Gizyn, 1994; Rasmussen *et al.*, 2001). As noted above, soil concentrations of uranium vary according to local geology. Although no single soil concentration can adequately represent the variance in background soil concentrations across Canada (Painter *et al.*, 1994), it is also essential to define a reasonable value for purpose of generic, national guidelines development. Refer to Table 5 for more details on background uranium concentrations in Canadian soils.

The total daily uranium intake via food was calculated using food intake rates for various age groups of Canadians (Table 15) and mean concentrations for 238-uranium determined in the 2001 UK Total Diet Study (Table 7). No Canadian food concentration data were located in the literature reviewed. However, given the similarities between the UK (238-U) and Ottawa (total U) adult dietary intakes, it was considered reasonable to assume that the UK data were applicable to Canada. Consumption of cereals-grains, sugar-sweets, and meat-poultry-eggs contributed the greatest to uranium exposure (in decreasing order) for all age groups (Table 16).

Table 17 summarizes total EDIs for uranium via all media for four age classes of the Canadian general population. It was estimated that adults, teenagers, school aged children, and toddlers were exposed to 1.6, 1.9, 1.7, and 1.3 µg U/day, respectively. Food constitutes the main source of uranium exposure, ranging from 77% (toddler, 7 months to 4 years) to 89% (teenagers, 12-19 years) of the total EDI. In the case of the toddler, drinking water and soil together comprise 22% of the total uranium exposure. These exposure rates correspond reasonably well with those provided by Taylor and Taylor (1997) where doses between 1 and 5 µg U/day were estimated for persons in uncontaminated regions of the U.S.

Certain Canadian subpopulations may be exposed to higher levels of uranium. Naturally occurring high levels of uranium in drinking water have been found in various locations in

Canada. Consumption of such waters would be the most likely route for higher Canadian exposure to uranium. Consumption of food washed or prepared with water containing high levels of uranium could also increase uranium exposure above the levels calculated in the present exposure analysis. In addition, people living near uranium mining areas could inhale higher concentrations of uranium from ambient air or from fugitive dust particles. Due to insufficient data, it is not possible, at this time, to perform an exposure assessment for those groups. However, the current analysis does suggest that, next to the consumption of food and water, the inhalation and incidental soil ingestion pathways are small contributors to total uranium exposure.

CHAPTER 6. DERIVATION OF ENVIRONMENTAL AND HUMAN HEALTH SOIL QUALITY GUIDELINES

Environmental Soil Quality Guidelines

Canadian soil quality guidelines are derived for the protection of receptors under four different land uses: agricultural, residential/parkland, commercial and industrial. The following derivation is based on the protocols described in CCME (2005).

All data selected for use in the following derivations have been screened for ecological relevance and are presented in preceding sections. Data with a soil pH below 4 were not selected for the purpose of soil quality guideline derivation. These data are considered outside the normal pH range of most soils in Canada. In addition, data were not selected if: soil pH was not recorded, no indication of soil texture was provided, inappropriate statistical analyses were used, the test was not conducted using soil or artificial soil, the test soil was amended with sewage sludge or a mixture of toxicants, or the test did not use controls.

Soil Quality Guidelines for Agricultural and Residential/Parkland Land Uses

Soil Quality Guideline for Soil Contact

The derivation of the soil quality guideline for soil contact (SQG_{SC}) is based on toxicological data for vascular plants and soil invertebrates. The toxicological data available for plants and invertebrates are presented in Chapter 4 and Tables 10 and 11. Data in the tables that are listed as "selected" were used in the guideline derivation, while those listed as "consulted" were not considered appropriate for use. A total of 7 studies were selected as acceptable for use, including 15 plant species and 4 invertebrate species. Endpoints from selected studies that were expressed as "greater than" (>) a certain concentration were not used to derive the guidelines, which reduced the number of represented plant species to 11. Effect concentrations spanned a large range depending on the study.

The minimum data requirements for use of the preferred weight-of-evidence approach for guideline derivation were met. Geometric means were calculated for similar endpoints reported for similar species including:

- LOEC and NOEC for germination in *Brassica rapa*
- LOEC and NOEC for stem length in *Brassica rapa*
- LOEC and NOEC for straw weight in *Brassica rapa*
- LOEC and NOEC for root length in northern wheatgrass
- EC_{20} for survival of *Onychiurus folsomi*
- EC_{20} for number of juveniles produced by *Onychiurus folsomi*
- EC_{20} for survival of *Folsomia candida*
- EC_{20} for number of juveniles produced by *Folsomia candida*

A total of 82 acceptable NOEC, LOEC, and EC20 values were used. The 25th percentile of the data distribution was calculated as 500 mg/kg (Figure 2).

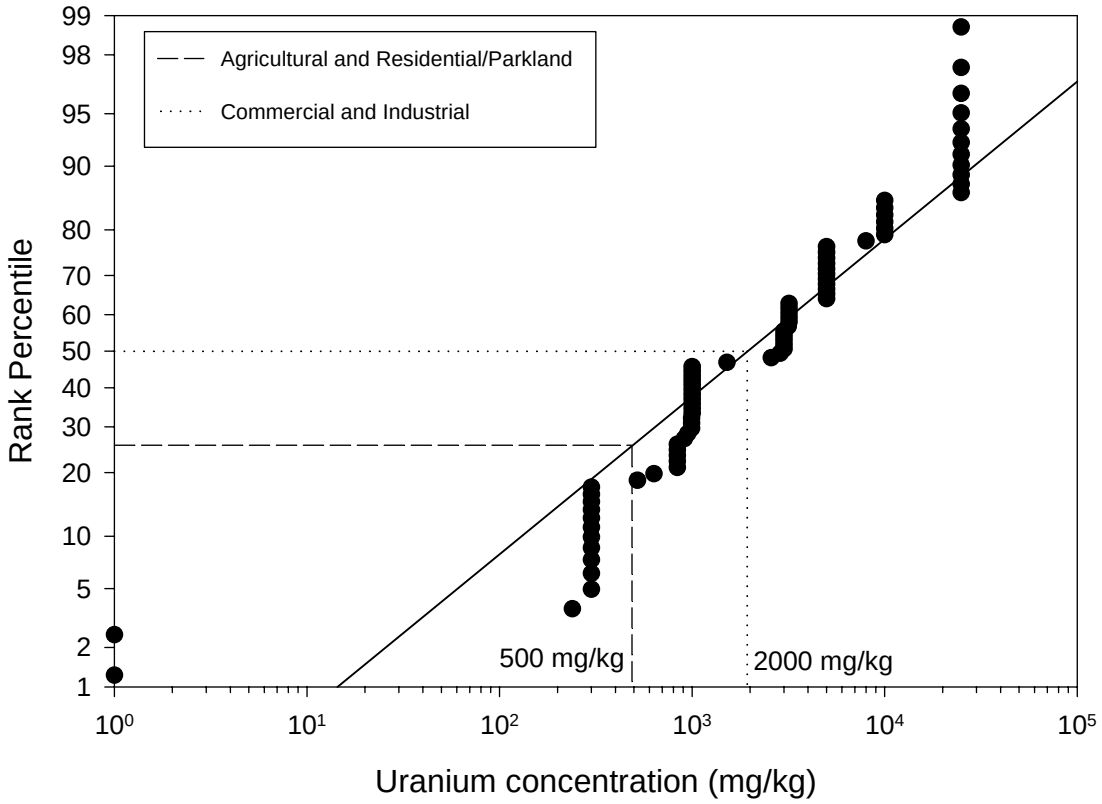

Figure 2. Soil contact toxicity data distribution for all land uses.

Nutrient and Energy Cycling Check

As discussed in CCME (2005), sulphatase and phosphatase enzyme activities vary in soils with phosphate and sulphate concentrations in the soil, and may be stabilized in soil outside the cell. Consequently, mineralization rates of phosphorus and sulphur are not good candidates for reporting on soil toxicity. The protocol for deriving soil quality guidelines (CCME 2005) recommends the use of data on nitrification, nitrogen-fixation, decomposition, respiration, or nitrogen mineralization. Therefore, phosphatase activity data reported by Sheppard *et al.* (1992) was not used in calculating a nutrient and energy cycling check. Several other studies on microbial processes described in Chapter 4 were not conducted in soil, so they too cannot be used to derive this check value. Acceptable data on microbial processes were only available from one study (Meyer *et al.* 1998a), therefore a nutrient and energy cycling check for uranium could not be calculated.

Groundwater Check for Aquatic Life

The groundwater check for the protection of aquatic life applies only to nonpolar organic compounds and not to metal contaminants. Therefore, this check was not calculated.
Thus, the SQG_{SC} for uranium is 500 mg/kg soil for agricultural and residential/parkland land uses.

Soil Quality Guidelines for Soil and Food Ingestion for Agricultural Land Uses

A soil quality guideline for soil and food ingestion (SQG_I) is only calculated for agricultural land uses. The following section provides the derivation of the SQG_I for domestic animals and wildlife that are primary consumers.

Development of Daily Threshold Effect Dose (DTED)

There were sufficient toxicological data to derive LOAELs to fulfill the minimum requirements for derivation of the SQG_I (Tables 6 and 7). The protocol (CCME 2005) requires a minimum of two mammalian oral studies (one of which should include a grazing herbivore such as an ungulate) and one avian oral study.

The database for avian species indicated a minimum NOAEL of 16 mg/kg bw/d for adverse effects in American black duck (Haseltine and Sileo 1983; Sample et al. 1996). Although single dose injection studies established lower LOAELs for Japanese quail (Kupsh et al. 1991) and Leghorn chickens (Harvey et al. 1986), this route of administration and the length of exposure were not considered relevant in the derivation of an environmental guideline.

For mammalian livestock, only one study on dairy cattle was available (fulfilling the requirement for a grazing ungulate). In this study, a LOAEL of 0.62 mg/kg bw/d was determined for effects on general health and milk yield of cows (Garner 1963).

All other oral toxicity data available for mammals were for laboratory animals such as mice, rats, rabbits and dogs (Table 7). The lowest LOAELs reported were 0.06 and 0.09 mg/kg bw/d for renal effects in male and female laboratory rats, respectively, that were administered uranyl nitrate in drinking water over a 91-day period (Gilman et al. 1998c). The next lowest acceptable LOAEL was 0.49 mg/kg bw/d for renal effects in female New Zealand white rabbits that were also administered uranyl nitrate in drinking water over a 91-day period (Gilman *et al.* 1998a). A lower LOAEL of 0.05 mg/kg bw/d was reported in the same study for renal effects in male rabbits (Gilman et al. 1998a). This endpoint, however, was considered suspect due to the potential confounding factor of a bacterial (*Pasturella multocida*) infection that was noted in several of the male rabbits.

Although laboratory results indicated that rats are more sensitive than rabbits, it was decided to use rabbits for the derivation of the SQG_I. Rabbits were considered more appropriate for several reasons. First, because rabbits are strictly herbivores, they are more representative of a grazing

species than rats, which are omnivorous. Second, when one considers the attributes of wild rats and wild rabbits (as opposed to laboratory strains), there is evidence to suggest that rabbits may be more sensitive. For example, based on wildlife daily food ingestion rates and body weights for rats and rabbits (U.S. EPA 1993; BCMELP 1996), ingestion rates corrected for body weight are higher for rabbits than for rats, at approximately 83 g dw food/kg bw/day versus 53 g dw food/kg bw/day, respectively. Also, the proportion of soil ingestion as a percentage of total dry matter intake has been estimated higher for rabbits, at 6.3% for black-tailed jackrabbits (Arthur and Gates 1988), than for rats, at 2.8% for cotton rats (Garten 1980 as cited in McMurter 1993). Although both of these species are native to the southern United States, it is expected that the proportion of soil ingestion for Canadian species of rabbits and rats would be comparable.

Therefore, rabbits were considered to be the appropriate organism upon which the SQG_I should be based.

Using the formula provided by CCME (2005):

DTED = lowest LOAEL/UF

where,

DTED = daily threshold effect dose (mg/kg bw/d)
LOAEL = lowest observed effect level (0.49 mg/kg bw/d) (Gilman *et al.* 1998a)
UF = uncertainty factor (1)

the DTED for mammalian species was calculated as 0.49 mg/kg bw/d.

No uncertainty factor was applied because there was an adequate amount of data available, representing several taxonomic groups, and the LOAEL was taken from a chronic study.

Receptor Parameters

The literature provided the following data for the receptor identified for consideration in the derivation of soil quality guidelines:

Receptor	Body Weight (kg)	Diet	Soil Ingestion Rate (kg/d dw)	Proportion of Soil Ingestion (as % of total Dry Matter Intake Rate)	Reference
Rabbit	1.2	100% vegetation	0.0097	2	U.S. EPA 1993; Arthur and Gates 1988

In the above table, body weight was determined from various studies cited in U.S. EPA (1993) that measured mean weights for adult rabbits. The diet of rabbits was noted to consist entirely of vegetation, with proportions of woody plants, grasses, and other herbaceous plants varying with

the season (U.S. EPA 1993). The soil intake rate and percentage of soil in the diet was obtained from Arthur and Gates (1988) who examined soil ingestion in rabbits.

Bioavailability

For the purpose of environmental quality guideline derivation, the bioavailability term is meant to represent the bioavailability of soil-sorbed uranium in rabbits. Gut-absorption of uranium forms that are tightly sorbed to soil particles is likely to be low; however this information was not available. Therefore, a conservative assumption was made that all uranium ingested in soil would be bioavailable, and thus a bioavailability factor of one was used.

Bioconcentration Factors

Numerous studies have looked at soil-to-plant bioconcentration of uranium (Table 8). By taking the geometric mean of all concentration ratios listed for plants in Table 8, a bioconcentration factor of 0.025 was obtained.

Determination of the Dry Matter Ingestion Rate

Arthur and Gates (1988) noted that the proportion of soil intake by rabbits is equal to 6.3% of their total dry matter intake rate (DMIR). Therefore, the DMIR can be calculated as:

$$DMIR = SIR \div PSI$$

where,

DMIR = dry matter ingestion rate (kg/day dw)
SIR = soil ingestion rate (0.0097 kg/day dw) (Arthur & Gates 1988)
PSI = percentage of soil in the diet, on a dry weight basis (6.3%) (Arthur & Gates 1988)

Therefore, the DMIR for rabbits is 0.154 kg/day dw.

Determination of the Rate of Food Ingestion

The rate of food ingestion is calculated as the proportion of the diet not consisting of soil, that is, the difference between the DMIR and the SIR, as follows:

$$FIR = DMIR - SIR$$

where,

FIR = food ingestion rate (kg/day dw)
DMIR = dry matter ingestion rate (0.154 kg/day dw) (calculated above)
SIR = soil ingestion rate (0.0097 kg/day dw) (Arthur & Gates 1988)

Therefore, the FIR for rabbits is 0.144 kg/day dw.

Calculation of the Soil Quality Guideline for Ingestion

As required by the CCME (2005) protocol, 25% of the DTED is apportioned for exposures via drinking water and dermal absorption. Therefore the total exposure via ingestion of food and soil should not exceed 75% of the DTED.

In the calculation of the soil quality guideline for ingestion, the exposure via soil ingestion is calculated via the combination of the SIR with the relevant bioconcentration factors along with the body weight, as follows:

Exposure via Soil Ingestion = SIR x SQG_I x BF / BW

where,

SIR = soil ingestion rate (kg/d dw)
SQG_I = soil quality guideline for soil and food ingestion (mg/kg)
BF = bioavailability factor (unitless)
BW = body weight (kg)

Sheppard and Evenden (1992b) found that uranium concentrations were higher in soil adhering to plant leaves than in bulk soil. This concentration enrichment is due to the greater sorption of uranium to clay particles. Clay tends to make up a higher proportion of the soil particles that are found adhered to plants than coarser particles such as sand (Sheppard and Evenden 1992b). Therefore, it is reasonable to expect that the proportions of soil ingested by rabbits from soil adhered to plants versus bulk soil will influence the concentration of uranium ingested. To consider this in the calculations, one would also need to know the differences between bioavailability of uranium in bulk soils versus plant-adhered soil. This information is not available, and could vary significantly on a site-specific basis.

In the calculation of the soil quality guideline for ingestion, the exposure via food ingestion is calculated via the combination of the FIR with the relevant bioconcentration factors and the body weight, as follows:

Exposure via Food Ingestion = FIR x SQG_I x BCF / BW

where,

FIR = food ingestion rate (kg/d dw)
SQG_I = soil quality guideline for soil and food ingestion (mg/kg)
BCF = soil to plant bioconcentration factor (unitless)
BW = body weight (kg)

These two equations can be combined and rearranged to solve for SQG_I, based on the assumption that the sum of the exposures via soil and food ingestion equals the exposure limit (0.75 x DTED):

$SQG_I = (0.75 \times DTED \times BW) / [(SIR \times BF) + (FIR \times BCF)]$
where,

SQG_I	= soil quality guideline for soil and food ingestion (mg/kg dw)
DTED	= daily threshold effects dose (0.49 mg/kg bw/d) (calculated above)
FIR	= food ingestion rate (0.144 kg/d dw) (calculated above)
SIR	= soil ingestion rate (0.0097 kg/d dw) (Arthur & Gates 1988)
BCF	= soil to plant bioconcentration factor (0.025 unitless) (calculated above)
BF	= bioavailability factor (unitless). This information was not available and thus a bioavailability factor of one is assumed.
BW	= body weight (1.2 kg) (U.S. EPA 1993)

Therefore, the calculated SQG_I for the receptor of concern is as follows:

Receptor	**SQG_I (mg/kg)**
rabbit	33

Therefore, the SQG_I for primary consumer mammalian species (rabbit) is 33 mg/kg for agricultural land use. Soil is estimated to contribute approximately three times as much uranium to the ingested dose as plant ingestion. The SQG_I is approximately 16 times the average background concentration of uranium in soil as reported by NCRP (1984) and is expected to be protective for sublethal histological changes in grazing mammals.

Summary and Selection of the SQG_E for Agricultural and Residential/Parkland Land Use

According to the CCME (2005) protocol, for agricultural land uses the lower of the SQG_I and the SQG_{SC} is adopted as the environmental soil quality guideline, which results in an SQG_E of 33 mg/kg. For residential/parkland land uses, the SQG_{SC} of 500 mg/kg is set as the SQG_E.

Soil Quality Guidelines for Commercial and Industrial Land Uses

Soil Quality Guideline for Soil Contact

The derivation of the soil quality guideline for soil contact (SQG_{SC}) is based on toxicological data for vascular plants and soil invertebrates. The toxicological data for plants and invertebrates is presented in Chapter 4 and Tables 10 and 11. As described earlier for agricultural and residential/parkland land uses, data were sufficient for use of the Weight-of-Evidence Method of guideline derivation. For commercial and industrial land uses, this involves calculating the Effects Concentration Low (ECL), i.e., the 50th percentile of the distribution of acceptable NOEC, LOEC, and EC_{20} values.

A total of 82 acceptable data points were available, ranging from 1 to 25 000 mg/kg (see Tables 10 and 11). Therefore, the ECL was calculated to be approximately 2000 mg/kg soil (Figure 2).

Nutrient and Energy Cycling Check

As discussed earlier for agricultural and residential/parkland land uses, it was not possible to complete a nutrient and energy cycling check for uranium.

Off-site Migration Check

When deriving soil quality guidelines for industrial sites, exposure scenarios consider only on-site exposure. Transfers of contaminated soil from one property to another is possible by environmental occurrences such as wind and water erosion (CCME 2005).

The Universal Soil Loss Equation and the Wind Erosion Equation are utilized to estimate the transfer of soil from one property to another. The following equation allows us to calculate the concentration (C_i) in eroded soil from the industrial site that will raise the contaminant concentration in the receiving soil to equal the residential/parkland guideline within a specific time frame. If the guideline for industrial sites is found to be above C_i, then neighbouring properties could potentially become unacceptably contaminated from off-site deposition (CCME 2005). The following equation has been derived to allow the calculation of C_i.

$$C_i = \frac{D_M \times C_M - \left(\left(D_M - D_D\right) \times BSC\right)}{D_D}$$

where,

C_i = concentration of contaminant in eroded soil (mg/kg)
D_M = depth of mixing, 2 cm (default)
C_M = concentration of contaminant in receiving soil after mixing, set equal to the soil quality guideline for residential/parkland land use (500 mg/kg)
D_D = depth of deposited material before mixing: 0.14 cm (assuming a deposition rate of 13.9 t/ha and bulk density for the eroded material of 1 t/m^3)
BSC = background concentration of contaminant in the receiving soil (2 mg/kg) (NCRP 1984)

Therefore, using the equations and assumptions above, the concentration of contaminant in the eroded soil was estimated to be 7116 mg/kg which is greater than the SQG_E of 2000 mg/kg for the industrial scenario. Therefore the industrial SQG_E is not changed.

It should be noted that with water or wind erosion, there may be some particle-sorting, with finer particles (e.g., clay) being eroded more than coarse particles (e.g., sand). The concentration of uranium sorbed to clay particles is high relative to other coarser particles, and therefore erosion can lead to concentration enrichment of uranium in eroded soil (Sheppard and Evenden 1992b). In a study that examined water-erosion of two soils with a 10% slope and contaminated with uranium, the mean uranium enrichment ratios observed for the silty clay loam soil and for the loam soil were 1.5 and 3.6, respectively (Sheppard and Evenden 1992b). Therefore, at the

industrial SQG_E of 2000 mg/kg, the concentration of uranium in the eroded soil could be 3.6 times higher, i.e., a concentration of 7200 mg/kg. This is very close to the modelled concentration of 7116 mg/kg that is considered protective of neighbouring residential soils. Therefore, even with concentration enrichment through particle-sorting, the SQG_E of 2000 mg/kg should protect against adverse effects through off-site migration.

Groundwater Check for Aquatic Life

The groundwater check for the protection of aquatic life applies only to nonpolar organic compounds and not to metal contaminants. Therefore, this check was not calculated.

Soil Quality Guideline for Soil and Food Ingestion

As discussed in the protocol (CCME 2005), the use of commercial and industrial sites by wildlife is considered to be greatly reduced, in comparison to that observed in agricultural scenarios. In addition, the normal land use activities on commercial and industrial sites do not depend on the maintenance of ecological functioning to the same degree. Therefore, as discussed in CCME (2005), soil contact is considered to represent the most significant pathway of exposure for ecological receptors under commercial/industrial land use. Because data do not permit the estimation of toxicity to wildlife via direct contact, it is assumed that the guidelines protective of soil invertebrates and plants would be protective of wildlife, based on differences in mobility and degree of direct soil contact.

Summary and Selection of the SQG_E for Commercial and Industrial Land Use

Based on the foregoing, for both commercial and industrial land uses the SQG_{SC} of 2000 mg/kg is selected as the SQG_E.

Data Gaps in the Derivation of Environmental Soil Quality Guidelines

There were several areas in which data were lacking, including the following:

- There were few studies of the effects of uranium on soil microorganisms and soil-dwelling invertebrates in the literature.
- In general, there were few data for the derivation of DTED for domestic livestock and terrestrial wildlife.
- There were no data on the bioavailability of uranium in soil to mammalian and avian species.

Human Health Soil Quality Guidelines

Estimated Daily Intakes

For the purpose of this exposure assessment, four age classes of the general population were considered: adults, teenagers, school aged children, and toddlers. Reference body weights and standard intakes of air, drinking water, and soil (including house dust) for each specified age class of the general population are presented in Table 17. The total daily uranium intake via food was calculated using food intake rates for various age groups of Canadians (Table 15) and mean concentrations for 238-uranium determined in the 2001 UK Total Diet Study (Table 7) (due to a lack of Canadian-specific data). It was assumed that the uranium concentrations determined in the UK Total Diet Study were representative of uranium levels in food in Canada. Dietary intake estimates for uranium are provided for all age groups in Table 16.

Daily intakes of uranium via air, drinking water, and soil were estimated by multiplying typical Canadian intake rates by the mean uranium concentrations occurring in the Canadian environment. In all cases, the average Canadian exposure has been estimated using uranium concentrations to which most Canadians are likely to be exposed based on the best information available at the present time.

Table 17 summarizes the daily intake estimates for uranium via all media for four age classes of the Canadian general population. The estimated daily intakes (EDI) for adults, teenagers, school aged children, and toddlers were 1.6, 1.9, 1.7, and 1.3 µg U/day, respectively. Food constitutes the main source of uranium exposure, ranging from 77% (toddler, 7 months to 4 years) to 89% (teenagers, 12-19 years) of the EDI. In the case of the toddler, drinking water and soil together comprise 22% of the total uranium exposure.

Exposure Limit for Human Receptors

Overall, the Health Canada TDI for uranium of 0.6 µg/kg bw/day was considered to be appropriate for use in the development of a Canadian $PSQG_{HH}$. In the cases of the ATSDR (1999) and U.S. EPA (2005a), these health agencies developed toxicity reference values that were considerably less conservative than the Health Canada value while WHO (2001a) offered a value that was only slightly more conservative (i.e., 0.5 µg/kg bw/day), and likely insignificantly so given the uncertainties in the determination of toxicological reference values. Finally, the WHO (2004a) offers a TDI value that is identical to Health Canada (1999). Overall, although there are uncertainties associated with this value, the Health Canada TDI of 0.6 µg/kg bw/day was considered to be the most appropriate value available for the purposes of establishing soil quality guidelines.

Soil Inhalation Rates

Soil inhalation rates were determined by multiplying air inhalation rates for a particular age group by the average soil particle concentration in air over a particular land use. Average air inhalation rates for toddlers (6 months to five years) and adults are 9.3 and 15.8 m^3/d, respectively (Richardson 1997; Health Canada 2003; CCME 2005). Health Canada (2003) reports dust concentrations of 0.76 µg/m^3 can be conservatively assumed (except in cases where regular vehicle traffic over unpaved soils or construction activities or other dust generating activities are expected). Consequently, the following soil inhalation rates were used:

Agricultural land use (toddler receptor)	=	7.1 µg/d or 7.1 x 10^{-9} kg/d
Residential/Parkland land use (toddler receptor)	=	7.1 µg/d or 7.1 x 10^{-9} kg/d
Commercial land use (toddler receptor)	=	7.1 µg/d or 7.1 x 10^{-9} kg/d
Industrial land use (adult receptor)	=	12 µg/d or 1.2 x 10^{-8} kg/d

Relative Absorption Factors

Relative absorption factors may be applied when the critical toxicological study has used a different medium than that under investigation, in order to account for the difference in

absorption of the contaminant by the body in the two different media. In this case, the critical study administered uranium in drinking water, so a relative absorption factor is needed to account for the difference in absorption of uranium in soil compared to uranium in water.

For ingestion of soil, a relative absorption rate of 1 (100%) was assumed. In estimation of a relative absorption factor, it has essentially been assumed that all uranium has an oral bioavailability equal to uranyl nitrate in the drinking water toxicological studies of Gilman *et al.* (1998 a;b;c). Although there may be information that suggests that certain uranium compounds may be more or less bioavailable than uranyl nitrate, no adequately specific information was identified that could be used in the development of the $PSQG_{HH}$. In addition, although there may be evidence available that indicates that ingestion absorption of uranium in soil would be lower than uranyl nitrate in drinking water (some unpublished *in vitro* studies have indicated that uranium in soil may be absorbed at only 70% of the rate that uranium in water may be absorbed from the gastrointestinal tract), it was considered that these data were insufficient to define an absorption value other than 100%.

In the case of uranium absorption across the respiratory tract (AF_L), the relative absorption rate was assumed to be 1 (100%). There is little information available to estimate how uranium adsorbed to soil particulates would be absorbed across the respiratory tract. Consequently, AF_L was assumed to be 1 (i.e., equal to the absorption rate in the oral toxicological studies). It is possible that the relative rate of uranium absorption across the respiratory tract could have conceivably been greater than across the gastrointestinal tract in the toxicological studies. Nevertheless, it is noted that the inhalation route is considered to be a relatively minor route of exposure and, thus, although this lack of route-specific data is not desirable, it is not expected to have a large impact on the data.

In the case of uranium absorption across the skin (AF_S), the relative absorption factor was assumed to be 0.05 (5%). There is little information available to estimate how uranium in soil would be absorbed across the skin. Some guidance suggests that a rate of 5% would be a reasonably conservative value (MDEP, 1995). Other guidance suggests lower values (e.g., ORNL [2005] recommends a value of 0.1%) as absolute absorption factors. However, it is important to note that the ORNL value is a generic default assumption for metals in general and is not specific for uranium. A variety of researchers contacted by these authors have suggested that this default assumption should not be applied to uranium. For the purposes of this assessment, a relative absorption factor of 0.05 (i.e., 5%) was used. It is noted that this was a relatively insensitive parameter in the development of the $PSQG_{HH}$.

It should be noted that Health Canada is considering dermal bioavailability studies of uranium in soil (M. Richardson, pers. comm.). Health Canada has initiated a research program conducting investigations of the dermal bioavailability of soil-borne contaminants using *in vitro* methods with intact, viable human skin. These will be the first such studies undertaken, to our knowledge. Methodological issues specific to uranium are being investigated to determine if dermal bioavailability studies of this element are feasible.

Agricultural and Residential/Parkland Land Uses (Toddler Receptor)

For determining an agricultural and a residential/parkland soil guideline it was assumed that the most appropriate receptor to use would be the preschool toddler due to a large exposure per unit mass. In accordance with the CCME guideline derivation procedures (CCME 2005), a preliminary soil quality guideline was derived for three exposure pathways (ingestion, inhalation, and dermal):

$$PSQG_{HH} = \frac{[(TDI - EDI) \times SF \times BW]}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S) \times ET_2] \times ET_1} + BSC$$

where,

$PSQG_{HH}$ = preliminary human health soil quality guideline (mg/kg)
TDI = tolerable daily intake (6.0×10^{-4} mg/kg bw/d) (Health Canada 1999)
EDI = estimated daily intake for the toddler (7.8×10^{-5} mg/kg bw/d) (based on various data – see Table 17)
SF = soil allocation factor of 20%, by default (CCME 2005)
BW = body weight for toddler (16.5 kg) (CCME 2005)
SIR = soil ingestion rate for toddler (8×10^{-5} kg/d) (CCME 2005)
IR_S = soil inhalation rate for toddler (7.1×10^{-9} kg/d) [i.e., inhalation rate for toddler = 9.3 m^3/d x suspended soil dust concentration of 7.6×10^{-10} kg/m^3 (Health Canada 2003)]
SR = soil dermal contact rate for toddler (6.9×10^{-5} kg/d) [hands surface area of 0.043 m^2 (CCME 2005) × soil adherence factor of 0.001 kg/m^2/d plus arms/legs surface area of 0.26 m^2 (CCME 2005) × soil adherence factor of 0.0001 kg/m^2/d (CCME 2005)]
BSC = background soil concentration (2 mg/kg) (NCRP 1984)
AF_G = relative absorption factor for soil: water in the gut (100%, assumed by default)
AF_L = relative absorption factor for soil: water in lung tissue (100%, by default)
AF_S = relative absorption factor for soil: water on skin (5%) (MDEP 1995)
ET_1 = exposure term 1 (unitless) – days per week/7 x weeks per year/52 at the site (1.0) [i.e., 7 days per week, 52 weeks per year assumed at the site (CCME 2005)]
ET_2 = exposure term 2 (unitless) – hours per day/24 at the site (1.0) [i.e., 24 hours per day assumed at the site (CCME 2005)]

The exposure term, ET_1 in the above equation, is the ratio of the defined exposure period for each land use to the maximum exposure period (24 hours/day x 365 days/year). Note that hours per day exposure (ET_2) is applied to soil inhalation, but is not considered for soil ingestion or dermal contact, consistent with Health Canada (2004) recommendations, since soil ingestion and dermal contact are not expected to occur at a uniform rate throughout the day.

The preliminary human health soil quality guideline ($PSQG_{HH}$) for uranium in agricultural soil was calculated to be 23 mg/kg. For residential/parkland land uses, the $PSQG_{HH}$ was calculated to be 23 mg/kg.

Commercial Land Use (Toddler Receptor)

Commercial land sites are generically defined as sites at which commercial activities predominate, such as a shopping mall. There are no manufacturing activities or residential sites present.

For threshold contaminants such as uranium the preschool toddler is assumed to be the most sensitive receptor. The commercial land use calculation is exactly the same as the Agricultural and Residential/Parkland calculations, the only differences being:

- exposure term 1 (ET_1) is 0.66 (based on 5 d/wk and 48 wk/y) due to the reduced amount of time the receptor spends on a commercial site.

- exposure term 2 (ET_2) is 0.42 (based on 10 h/d) due to the reduced amount of time the receptor spends on a commercial site.

The preliminary human health soil quality guideline ($PSQG_{HH}$) for uranium on commercial lands was calculated as 33 mg/kg.

Industrial Land Use (Adult Receptor)

In an industrial scenario, occupational exposure will be the primary route of exposure, hence the use of an adult receptor. Exposure for an adult at an industrial site is assumed to be 10 h/d, 5 d/wk and 48 wk/y. Examples of industrial lands could be manufacturing plants.

$$PSQG_{HH} = \frac{[(TDI - EDI) \times SF \times BW]}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S) \times ET_2] \times ET_1} + BSC$$

where,

$PSQG_{HH}$	= preliminary human health soil quality guideline (mg/kg)
TDI	= tolerable daily intake (6.0 x 10^{-4} mg/kg bw/d) (Health Canada 1999)
EDI	= estimated daily intake for adult (2.3 x 10^{-5} mg/kg bw/d) (based on various data – see Table 17)
SF	= soil allocation factor of 20%, by default (CCME 2005)
BW	= body weight for adult (70.7 kg) (CCME 2005)
SIR	= soil ingestion rate for adult (2 x 10^{-5} kg/d) (CCME 2005)
IR_S	= soil inhalation rate for adult (1.2 x 10^{-8} kg/d) [i.e., inhalation rate of 15.8 m^3/d x suspended soil dust concentration of 7.6 x 10^{-10} kg/m^3 (Health Canada 2004)]
SR	= soil dermal contact rate for adult (1.1 x 10^{-4} kg/d) [hands surface area of 0.089 m^2 (CCME 2005) × soil adherence factor of 0.001 kg/m^2/d plus arms surface area of 0.25 m^2 (CCME 2005) × soil adherence factor of 0.0001 kg/m^2/d (CCME 2005)]
BSC	= background soil concentration (2 mg/kg) (NCRP 1984)
AF_G	= relative absorption factor for soil: water in the gut (100%, assumed by default)
AF_L	= relative absorption factor for soil: water in lung tissue (100%, by default)
AF_S	= relative absorption factor for soil: water on skin (5%) (MDEP 1995)

ET_1 = exposure term 1 (unitless) – days per week/7 x weeks per year/52 at the site (0.66) [i.e., 5 days per week, 48 weeks per year assumed at the site (CCME 2005)]

ET_2 = exposure term 2 (unitless) – hours per day/24 at the site (0.42) [i.e., 10 hours per day assumed at the site (CCME 2005)]

Using this equation, the preliminary human health soil quality guideline ($PSQG_{HH}$) for uranium on industrial land was calculated as 510 mg/kg.

Guideline for Protection of Groundwater

No guideline for protection of groundwater was derived for uranium due to restrictions on the mathematical model when applied to metals (Nason 1995).

Off-site Migration Check

When deriving soil quality guidelines for industrial sites, exposure scenarios consider only on-site exposure. Transfers of contaminated soil from one property to another are possible by environmental occurrences such as wind and water erosion (CCME 2005).

The Universal Soil Loss Equation and the Wind Erosion Equation are utilized to estimate the transfer of soil from one property to another. The following equation allows us to calculate the concentration (C_i) in eroded soil from the industrial site that will raise the contaminant concentration in the receiving soil to equal the residential/parkland guideline within a specific time frame. If the guideline for industrial sites is found to be above C_i, then neighbouring properties could potentially become unacceptably contaminated from off-site deposition (CCME 2005). The following equation has been derived to allow the calculation of C_i.

$$C_i = \frac{D_M \times C_M - \left(\left(D_M - D_D\right) \times BSC\right)}{D_D}$$

where,

C_i = concentration of contaminant in eroded soil (mg/kg)

D_M = depth of mixing, 2 cm (default)

C_M = concentration of contaminant in receiving soil after mixing, set equal to the soil quality guideline for residential/parkland land use (23 mg/kg)

D_D = depth of deposited material before mixing: 0.14 cm (assuming a deposition rate of 13.9 t/ha and bulk density for the eroded material of 1 t/m^3

BSC = background concentration of contaminant in the receiving soil (2.0 mg/kg) (NCRP 1984)

Therefore, using the equations and assumptions above, the concentration of contaminant in the eroded soil was estimated to be 300 mg/kg which is less than the $PSQG_{HH}$ of 510 mg/kg for the industrial scenario. Therefore, the industrial $PSQG_{HH}$ should be set to 300 mg/kg.

Discussion of Uncertainties Associated with the $PSQG_{HH}$

The $PSQG_{HH}$ provided in this section are felt to be protective of human health at most sites. Some of the issues most important to the analysis and development of the uranium $PSQG_{HH}$ are described below.

To determine an acceptable level of exposure to uranium for development of $PSQG_{HH}$, the scientific positions of Health Canada (i.e., TDI = 0.6 µg/kg bw/day), WHO (i.e., TDI = 0.5 to 0.6 µg/kg bw/day), U.S. EPA (i.e., RfD = 3 µg/kg bw/day) and U.S. ATSDR (i.e., MRL = 2 µg/kg bw/day) were considered. The toxicological database for uranium is not as great as desirable and there is a moderate degree of uncertainty in the toxicity assessment. For the purposes of this assessment, the Health Canada TDI of 0.6 µg/kg bw/day was used. This TDI is similar to that recommended by WHO (2004a), slightly greater than WHO (2001a) and substantially lower than those used by ATSDR (1999) and U.S. EPA (2005a). All of these international health agencies provide values for use in human health risk assessment that are intended to protect the general public from exposures to uranium.

In order to assist in the interpretation of potential health risks, exposures that may result from the $PSQG_{HH}$ were compared to other benchmarks of exposure and potential health effects. Based on procedures described in this document, a toddler exposed to a uranium soil concentration of 22 mg/kg at a residence (24 hours per day, 7 days per week, 52 weeks per year) would have an estimated exposure rate of about 0.1 µg/kg bw/day which is about equivalent to the exposure that young children would receive from the typical background food supply (see Tables 16 and 17). In addition, exposures from soils with concentrations equal to the residential $PSQG_{HH}$ of 22 mg/kg would result in exposures that are 16% of the Health Canada TDI, 3% of the U.S. EPA RfD and 5% of ATSDR MRL.

The threshold concentration for renal injury in humans has been conservatively estimated to be 1 mg/kg renal tissue (Wrenn *et al.* 1985). Bosshard *et al.* (1992) reviewed toxicological data and proposed a NOEL of 1 mg/kg bw/d based on a dog feeding study concerning nephrotoxic effects. Morris and Meinhold (1995; 1998) provide a probabilistic model that estimates exposures in the range of those from the $PSQG_{HH}$ (i.e., 0.1 µg/kg bw/day or less) would likely be associated with kidney concentrations less than 0.01 mg/kg (i.e., more than 2 orders of magnitude lower than the tissue concentration of 1 mg/kg that has been discussed as a threshold level for nephrotoxicity by Wren *et al.* [1985] and Kocher [1989]). Similarly, Hakonson-Hayes *et al.* (2002) provide estimates for drinking water that would suggest the even lower dose rates from $PSQG_{HH}$ would result in kidney uranium concentrations less than the 0.01 mg/kg.

As shown in the equation provided in this document, the soil ingestion route is the dominant pathway such that uncertainties that may be associated with various aspects of dermal and inhalation exposures are unlikely to drive a risk assessment. In the case of dermal exposure, it is felt that dermal absorption of uranium would typically be a relatively minor exposure pathway (as compared to soil ingestion). In the case of inhalation of dusts, this pathway was evaluated using the Health Canada TDI. However, even if the most stringent ATSDR MRL was considered

(i.e., MRL of 0.3 μgm^3 for soluble uranium), the dust concentrations expected from sites at the $PSQG_{HH}$ would be much less than any value of concern (e.g., an industrial site with a $PSQG_{HH}$ of 280 mg/kg and a particulate concentration of 0.76 $\mu g/m^3$ would have a uranium concentration of 0.00021 $\mu g/m^3$). At sites where unusually high dust suspension is observed (e.g., construction activities and/or where there is excessive vehicular traffic on dirt roads), a lower value may need to be considered on a site-specific basis.

With respect to the soil ingestion route, the oral biovailability of uranium in soil was essentially assumed to equal the bioavailability of uranium in drinking water (as uranyl nitrate). As noted earlier, no definitive data were available such that the assessment did not quantitatively account for the potential reduced bioavailability of uranium in soil.
Although localized areas within Canada may have different background concentrations, the $PSQG_{HH}$ developed for uranium should be protective of most situations. Drinking water and air concentrations contributed relatively small amounts to the EDI calculations (i.e., < 7% of the total EDI) such that variations from the assumed average Canadian concentrations will typically have only a minor impact on the EDI and $PSQG_{HH}$. In the case of soil, it is noted that the uranium background soil concentration (BSC) was assumed to be 2 mg/kg for estimating exposure of the general Canadian population to uranium in soil. This was considered to be a reasonable value based on the available data.

Although the $PSQG_{HH}$ are felt to be protective at most sites, certain exposure pathways have not been evaluated in the development of the $PSQG_{HH}$. More specifically, the $PSQG_{HH}$ have not evaluated garden produce consumption or drinking water consumption. In the case of garden produce consumption, it is noted that this pathway has not been evaluated in the development of the $PSQG_{HH}$ provided above. At sites where appreciable amounts of garden produce are consumed, a lower value may need to be considered. In the case of drinking water consumption, it is noted that this pathway has not been evaluated in the development of the $PSQG_{HH}$ provided above. At sites where drinking water is sourced from nearby wells, a lower value may need to be considered.

As a result, the $PSQG_{HH}$ derived herein should be considered to be conservative even though uncertainties with some data exist. Nevertheless, as new toxicological and other data become available, the $PSQG_{HH}$ should be re-evaluated to ensure adequate protection of human health. With the above in mind, the $PSQG_{HH}$ are felt to be protective of human health at most sites.

Land Use	**$PSQG_{HH}$ (mg/kg)**
Agricultural	23
Residential/Parkland	23
Commercial	33
Industrial *	300 (due to off-site migration check)

* Guidelines for industrial land cannot result in contamination of abutting residential properties through surface erosion, above the residential guideline value; a model that considers surface erosion from the industrial property to the abutting residential property is employed to 'check' for this potential.

CHAPTER 7. RECOMMENDED CANADIAN SOIL QUALITY GUIDELINES

According to the CCME protocol (CCME 2005), both environmental and human health soil quality guidelines are developed for four land uses: agricultural, residential/parkland, commercial, and industrial. The lowest value generated by the two approaches for each of the four land uses is recommended by CCME as the Canadian Soil Quality Guideline. Therefore, the recommended Canadian Soil Quality Guidelines for the protection of environmental and human health are 23 mg/kg for agricultural land use, 23 mg/kg for residential/parkland land use, 33 mg/kg for commercial land use, and 300 mg/kg for industrial land use. Table 18 summarizes the soil quality guideline values derived for all exposure pathways and land uses utilized in the determination of the Canadian Soil Quality Guidelines for uranium.

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TABLES

Table 1. Physical and Chemical Properties of Uranium.

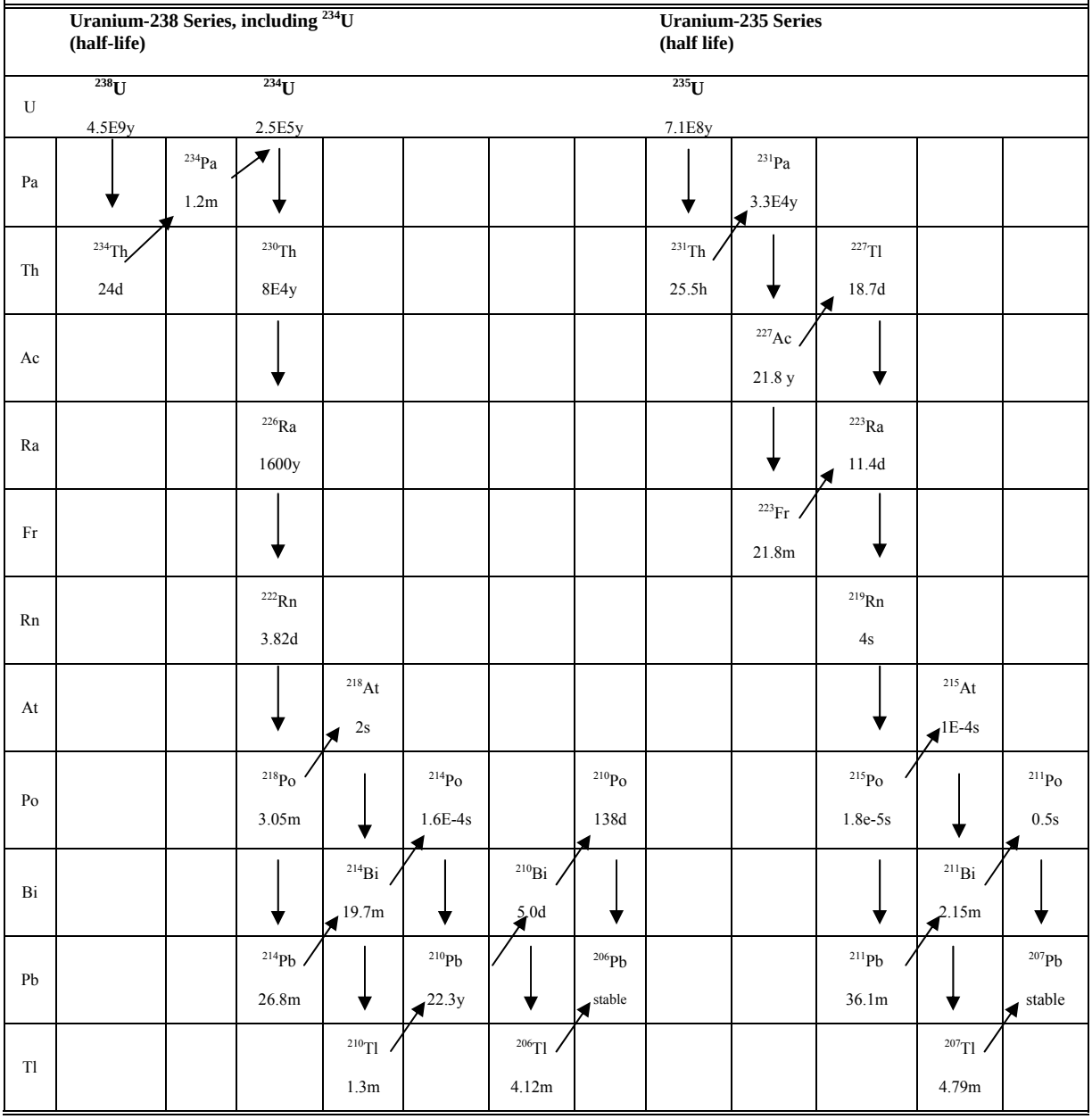
Property	Value	References
CAS number	7440-61-1	
Molecular formula	U^{2+}, U^{3+}, U^{4+}, U^{5+}, U^{6+}	Weast and Astle 1982
Isotopes	^{234}U(0.006 %) ^{235}U(0.72 %) ^{238}U(99.27 %)	Weast and Astle 1982
Atomic number	92	Merck Index 1989
Electron structure	$1s^2$, $2s^2$, $2p^6$, $3s^2$, $3p^6$, $3d^{10}$, $4s^2$, $4p^6$, $4d^{10}$, $4f^{14}$, $5s^2$, $5p^6$, $5d^{10}$, $5f^3$, $6s^2$, $6p^6$, $6d^1$, $7s^2$	Weast and Astle 1982
Chemical structure	Body-centred cubic	Merck Index 1989
Relative molecular mass	238.0289	Weast and Astle 1982
Melting point	1132.3 °C	Weast and Astle 1982
Boiling point	3818 °C	Weast and Astle 1982
Vapour pressure	1 mmHg at 2450 °C	Weast and Astle 1982
Heat of vaporization	446.7 kJ/mol	Merck Index 1989
Specific gravity	18.95	Weast and Astle 1982

Table 2. Occurrence and Radioactive Properties of Natural Uranium

	% of Total Uranium in Crustal rock			
Isotope	**By weight**	**By radioactivity**	**Half-life (years)**	**Alpha energies-MeV (abundance)**
^{234}U	0.0055	48.9	$2.45x10^{5}$	4.776 (72.5%)
				4.723 (27.5%)
^{235}U	0.720	2.2	$7.04x10^{8}$	4.597 (5%)
				4.395 (55%)
				4.370 (6%)
				4.364 (11%)
				4.216 (5.7%)
				Others (17.3%)
^{238}U	99.2745	48.9	$4.46x10^{9}$	4.196 (77%)
				4.147 (23%)

Source: ATSDR 1999.

Table 3. Naturally Occurring Uranium Isotope Decay Series



source: ATSDR 1999
s-seconds
m-minutes
d-days
y-years

alpha decay

beta decay

Table 4. Analytical Methods for Determining Total Uranium in Environmental Samples

Sample Matrix	Sample Preparation	Analytical Method	Sample Detection Limit	Accuracy	Reference
Air	Air particulate collection on glass fiber filter; digestion in HNO_3	ICP-MS	0.1 µg/L in final solution	No data	Boomer and Powell 1987
Air	Collection on cellulose filters	INAA	0.03 µg filter	No data	Querol *et al.* 1997
Drinking water	Direct Analysis	Laser-induced Fluorescence	0.2 µg/L	±3% at 30 µg/L level	Horvarth 2003
Drinking water	Direct analysis or concentration by coprecipitation and solvent extraction; fusion	Fluorometry	<20 µg/L (direct) 0.1 µg/L (cleaned)	104% cleaned	Krieger and Whittaker 1980 (EPA Method 908.1)
Drinking water	Sample chelation in EDTA; addition of Fluron	Laser-induced fluorometry	0.08 µg/L	100% at 1 µg/L	Velten and Jacobs 1984 (EPA Method 908.2)
Natural waters	Sample concentration by cation-exchange resin, separation by ion-exchange resin and complexation with Arsenazo III	Spectrophotometry	0.1 µg/L	80%	Paunescu 1986
Water	Preconcentration using the ion exchanger Hyphan	ICP α-spectrometry	2 ppb 0.4 ppb	No data	Van Britsom *et al.* 1995
Water	Sample fusion with NaF and LiF	Fluorometry	5 µg/L	117.5% at 6.3 µg/L	ASTM 1986 (Method D2907-83)
Water	Preconcentration by complexation with oxine and adsorption on activated carbon	NAA	3 µg/L	>80%	Holzbecher and Ryan 1980
Water	Extraction by ion-exchange; dissolution in low oxygen solvent; irradiation	Delayed neutron analysis	0.4 µg/L	No data	Zielinski and McKown 1984
Water	Wet-ashed; reaction with complexant	Pulsed-laser phosphorimetry	0.05 ppb	103% (average)	ASTM 1994 (Method 5174-91)
Groundwater	Separation and concentration on two HPLC columns; complexation with Arsenazo III	Spectrophotometry	1-2 µg/L	No data	Kerr *et al.* 1988
Water and wastes	Acid digestion; filtration (dissolved); acid digestion (total recoverable)	ICP-MS	0.1 µg/L	105-110%	Long and Martin 1991 (EPA Method 200.8)
Soil	Dissolution in HCl-HNO_3-HF; purification by coprecipitation, solvent extraction and electrodeposition	α-spectrometry (isotope quantification)	0.03 µg/sample	67%	Singh and Wrenn 1988
Soil	Digestion with HCl-HNO3	AAS GFAAS ICP-AES ICP-MS	Depends on analytical method	Depends on analytical method	BCMELP 2001
Soil, sediment, and biota	Ashing; fusion with KF and $K_2S_2O_7$; purification by extraction with triisooctylamine, anion exchange chromatography and coprecipitation.	α-spectrometry (isotope quantification)	No data	No data	EPA 1984
Soil, sediment, and biota	Ashing; extraction into triisooctylamine, strip from triisooctylamine with HNO_3 and coprecipitation with lanthanum.	α-spectroscopy	No data	No data	EPA 1984

Table 4. Analytical Methods for Determining Total Uranium in Environmental Samples Cont'd

Sample Matrix	Sample Preparation	Analytical Method	Sample Detection Limit	Accuracy	Reference
Vegetation	Sample dried and homogenized; dry and wet ashing	ICP-MS	0.1 µg/L in final solution	No data	Boomer and Powell 1987
Vegetation	Sample dried and homogenized; wet ashing and purification by solvent extraction	Laser Fluorometry (total soluble U)	0.05 mg/kg in plant ash	No data	Harms *et al.* 1981

Source: ATSDR 1999 (with the addition of Horvarth 2003; BCMELP 2001, and; Van Britsom *et al.* 1995).

AAS - Atomic Absorption Spectrometry
GFAAS - Graphite Furnace Atomic Absorption Spectrometry
ICP-AES - Inductive Coupled Plasma Atomic Emission Spectrometry
ICP-MS - Inductive Coupled Plasma Mass Spectrometry
NAA - Neutron Activation Analysis
INAA – Instrumental Neutron Activation and Analysis

Table 5. Background Concentrations of Uranium in Canadian Surface Soil

Location	Number of samples	Soil Concentration Mean (mg/kg) (range)	Analytical Technique	Reference
NEW BRUNSWICK				
East St. John (urban)	18	1.9	ICP-MS	Pilgrim and Schroeder 1997
West St. John (urban)	4	1.8		
Fredericton area (rural)	2	2.3		
ONTARIO				
Old urban parkland	60	1.9[a] (0.8^{b} to 2.2^{c})	HNO_3 digestion ICP-MS	OMEE 1993
Rural parkland	84	2.1[a] (2.0^{b} to 9.7^{c})		
Windsor				
Urban	18	0.9 (0.4 to 1.3)	HNO_3 digestion ICP-MS	Gizyn 1999
Rural	12	1.2 (0.8 to 2.2)		
Ottawa Urban garden soil	50	1.17 (0.66 to 2.64)	ICP-MS	Rasmussen *et al.* 2001
South March Uranium deposit Background site	NR	1.9	NR	Gordon 1992
Port Hope Uranium deposit Background site (region specific)	NR	33	NAA	Stantec 2004
NORTHERN MANITOBA	6	0.43 to 1.29	ICP-MS	E. Yee, Manitoba Conservation, pers. comm.
NORTHERN SASKATCHEWAN				
Key Lake Uranium deposit	3	3.5	DNC-slowpoke II reactor	Thomas 2000
Background sites: Peat layer in bog Jackpine habitat	3	1.1		
BRITISH COLUMBIA				
Prairie Flats Uranium deposit Background site	NR	3	Fluorimetry	Van Netten and Morley 1982b
Blizzard Uranium deposit Background site	NR	3 to 7	NR	Gordon 1992

NR – not reported
[a] 98^{th} percentile
[b] lower concentration limit
[c] upper concentration limit
ICP-MS - Inductive Coupled Plasma Mass Spectrometry
NAA - Neutron Activation Analysis
DNC – Delayed Neutron Counting

Table 6. Mean Uranium Content (mg/kg) in New York City Food Samples (1978)[1]

Food Group	Total U (mBq/kg)[2]	Total U (mg/kg)
FRESH VEGETABLES	48.5	0.0019
CANNED VEGETABLES	8.79	0.0003
ROOT VEGETABLES	20.3	0.0008
POTATOES	2.03	0.0001
DRY BEANS	60.03	0.0024
FRESH FRUIT	4	0.0002
CANNED FRUIT	3.69	0.0001
FRUIT JUICES	1.15	0.0000
BAKERY PRODUCTS	53.98	0.0021
FLOUR	9.75	0.0004
WHOLE GRAIN PRODUCTS	43.23	0.0017
MACARONI	7.44	0.0003
RICE	5.65	0.0002
MEAT	4.18	0.0002
POULTRY	1.62	0.0001
EGGS	2.83	0.0001
FISH	31.12	0.0012
SHELLFISH	4225	0.1677
DAIRY PRODUCTS	1.78	0.000071

[1]Fisenne *et al.* 1987.
[2]Assumed 1 μg/g = 25.2 mBq/kg ÷ 1000 (Thomas and Gates 1999)

Table 7. Mean 238U Content (mg/kg) in United Kingdom Food Samples (2001)[1]

Food Group	Mean ^{238}U mg/kg
BREAD	0.0035
MISCELLANEOUS CEREALS	0.0018
CARCASS MEAT	<0.00006
OFFAL	0.0012
MEAT PRODUCTS	0.0023
POULTRY	0.0005
FISH	0.0035
OILS AND FATS	(0.0006)
EGGS	(0.00009)
SUGARS AND PRESERVES	0.0012
GREEN VEGETABLES	0.0004
POTATOES	0.0005
OTHER VEGETABLES	0.0005
CANNED VEGETABLES	0.0012
FRESH FRUIT	0.0001
FRUIT PRODUCTS	0.0003
BEVERAGES	(0.00004)
MILK	(0.00002)
DAIRY PRODUCE	0.0016
NUTS	0.0007

[1]UK FSA, 2005
< indicates minimum value below limit of detection
( ) indicates values below limit of quantification but above the limit of detection

Table 8. Concentration Ratios for Uranium in Various Species

Organism	Concentration Ratio (geometric mean with range in brackets)[a]	Reference
Plants		
***Abelmoschus esculentus* (okra)**	**0.003[b] (0.0008 – 0.0082)**	**Lakshmanan and Venkateswarlu 1988**
***Allium cepa* (onion)**	**0.311 (0.213 - 0.455)**	**Tracy et al. 1983**
***Alnus rugosa* (speckled alder)**	**0.036**	**Sheppard and Thibault 1984**
***Alnus* sp. (alder)**	**0.01**	**Lopatkina et al. 1970**
***Arctium* sp. (burdock)**	**0.021**	**Sheppard et al. 2004**
***Asclepias* sp. (milkweed)**	**0.046 (0.033 – 0.065)**	**Sheppard et al. 2004**
***Asparagus officinalis* (asparagus)**	**0.260**	**Tracy et al. 1983**
***Aster* sp. (asters)**	**0.011 (0.0085 – 0.013)**	**Sheppard et al. 2004**
***Avena sativa* (oat)**	**0.027 (0.001 – 0.76)**	**Van Netten and Morley 1982a**
***Beta vulgaris* (chard)**	**0.009**	**Sheppard et al. 1989**
	0.547 (0.13 – 2.3)	**Shahandeh and Hossner 2002**
***Beta vulgaris* (beet)**	**0.812 (0.161 – 4.1)**	**Tracy et al. 1983**
	0.005 (0.002 - 0.009)	**Sheppard et al. 1989**
	0.906 (0.20 – 4.1)	**Shahandeh and Hossner 2002**
	0.0057 (0.0015 - 0.022)	**Sheppard et al. 2004**
***Betula nana* (shrubby birch)**	**0.107 (0.02 – 0.3)**	**Lopatkina et al. 1970**
***Betula papyrifera* (paper birch)**	**0.030**	**Sheppard and Thibault 1984**
***Betula* sp. (birch)**	**0.016 (0.002 – 0.4)**	**Lopatkina et al. 1970**
***Brassica juncea* (brown mustard)**	**0.616 (0.19 – 2.0)**	**Shahandeh and Hossner 2002**
***Brassica juncea* (Indian mustard)**	**1.094 (0.29 – 5.4)**	**Shahandeh and Hossner 2002**
***Brassica juncea* (Mustard green)**	**0.894 (0.20 – 4.0)**	**Shahandeh and Hossner 2002**
***Brassica oleracea var. italica* (broccoli)**	**0.005**	**Sheppard et al. 1989**
***Brassica perkinensis* (Chinese cabbage)**	**0.747 (0.18 – 3.1)**	**Shahandeh and Hossner 2002**
***Brassica rapa* (turnip)**	**0.767 (0.21 – 2.8)**	**Shahandeh and Hossner 2002**
***Capsicum* sp. (pepper)**	**0.004 (0.0026 – 0.0067)**	**Sheppard et al. 2004**
***Cucumis sativus* (cucumber)**	**0.012**	**Tracy et al. 1983**
	0.0009	**Sheppard et al. 1989**
	0.0017 (0.001 – 0.0029)	**Sheppard et al. 2004**
***Cucurbita sp.* (pumpkin, squash)**	**0.133**	**Tracy et al. 1983**
	0.0006 (0.00031 – 0.0011)	**Sheppard et al. 2004**
***Cynodon dactylon* (bermudagrass)**	**0.274 (0.05 – 1.5)**	**Shahandeh and Hossner 2002**
***Daucus carota* (carrot)**	**0.305 (0.052 – 1.576)**	**Tracy et al. 1983**
	0.006	**Sheppard et al. 1989**
	(0.0043 – 0.052)	**Sheppard et al. 2004**
***Equisetum arvense* (horsetail)**	**0.046**	**Sheppard et al. 2004**
***Fragaria* sp. (strawberry)**	**0.261**	**Tracy et al. 1983**
***Fraxinus pennsylvanica* (green ash)**	**0.028**	**Sheppard et al. 2004**
***Gleochoma hederacea* (gill-over-the-ground)**	**0.063**	**Sheppard et al. 2004**
***Helianthus annus* (sunflower)**	**1.754 (0.38 – 8.1)**	**Shahandeh and Hossner 2002**
***Hordeum vulgare* (barley)**	**0.170 (0.020 – 0.433)**	**Van Netten and Morley 1982b**
	0.002	**Sheppard et al. 1989**
***Impatiens capensis* (jewelweed)**	**0.0069**	**Sheppard et al. 2004**
***Juniperus communis* (common juniper)**	**0.037 (0.024 – 0.056)**	**Sheppard and Thibault 1984**
***Lactuca sativa* (lettuce)**	**0.025**	**Sheppard et al. 1989**
***Lagenaria leucantha* (bottle gourd)**	**0.001[b] (0.0003 – 0.0034)**	**Lakshmanan and Venkateswarlu 1988**
***Larix* sp. (larch)**	**0.067 (0.03 – 0.2)**	**Lopatkina et al. 1970**
***Ledum groenlandicum* (Labrador tea)**	**0.119 (0.05 – 0.368)**	**Sheppard and Thibault 1984**
***Ledum* sp. (Labrador tea)**	**0.016 (0.001 – 0.2)**	**Lopatkina et al. 1970**
***Lonicera oblongifolia* (swamp fly honeysuckle)**	**0.003 (0.0006 – 0.016)**	**Sheppard et al. 2004**
***Lycopersicon esculentum* (tomato)**	**0.063 (0.01 – 0.4)**	**Tracy et al. 1983**
	0.00056	**Sheppard et al. 2004**
***Lycopodium annotinum* (stiff club moss)**	**0.316**	**Sheppard and Thibault 1984**
***Malus* sp. (apple)**	**0.320 (0.293 – 0.35)**	**Tracy et al. 1983**
	0.005 (0.00081 – 0.036)	**Sheppard et al. 2004**
***Nepeta cataria* (catnip)**	**0.022**	**Sheppard et al. 2004**
***Panicum virgatum* (switchgrass)**	**0.338 (0.06 – 1.9)**	**Shahandeh and Hossner 2002**
***Parthenocissus inserta* (Virginia creeper)**	**0.004 (0.00053 – 0.021)**	**Sheppard et al. 2004**
***Phaseolus* sp. (bean)**	**0.048 (0.035 – 0.066)**	**Sheppard & Evenden 1992a**
	0.091 (0.055 – 0.15)	**Tracy et al. 1983**
***Picea mariana* (black spruce)**	**0.050 (0.012 – 0.316)**	**Sheppard and Thibault 1984**
***Pinus banksiana* (jack pine)**	**0.066 (0.022 – 0.181)**	**Sheppard and Thibault 1984**
***Pinus sylvestris* (Scots pine)**	**0.422 (0.03 – 6.1)**	**Sheppard et al. 1985**
***Pleurozium schreberi* (fern moss)**	**0.254**	**Sheppard and Thibault 1984**

Table 8. Concentration Ratios for Uranium in Various Species Cont'd

Organism	Concentration Ratio (geometric mean with range in brackets)[a]	Reference
***Poa* sp. (grass)**	**0.014 (0.0069 – 0.034)**	**Sheppard et al. 2004**
***Pyrus* sp. (pear)**	**0.00041**	**Sheppard et al. 2004**
***Raphanus sativus* (radish)**	**0.016 (0.001 – 0.5)**	**Van Netten and Morley 1983**
	0.043 (0.013 – 0.237)	**Sheppard & Evenden 1992a**
	0.032 (0.002 – 0.095)	**Sheppard et al. 1989**
	0.014	**Sheppard et al. 1989**
	0.003[b] (0.0006 – 0.0132)	**Lakshmanan and Venkateswarlu 1988**
***Rhamnus* sp. (buckthorn)**	**0.001 (0.00032 – 0.0067)**	**Sheppard et al. 2004**
***Rheum rhabarbarum* (rhubarb)**	**0.200 (0.08 – 0.5)**	**Tracy et al. 1983**
***Rhus typhina* (sumac)**	**0.011 (0.0057 – 0.021)**	**Sheppard et al. 2004**
***Robinia pseudo-acacia* (black locust)**	**0.017**	**Sheppard et al. 2004**
***Rubus* sp. (raspberry)**	**0.468 (0.26 – 1.8)**	**Tracy et al. 1983**
	0.022 (0.019 – 0.025)	**Sheppard et al. 2004**
***Salix* sp. (willow)**	**0.316 (0.1 – 1.0)**	**Lopatkina et al. 1970**
***Smilacina stellata* (false Solomon's seal)**	**0.0076**	**Sheppard et al. 2004**
***Solanum melongena* (eggplant)**	**0.001[b] (0.0004 – 0.0039)**	**Lakshmanan and Venkateswarlu 1988**
***Solanum tuberosum* (potato)**	**0.126 (0.02 – 0.8)**	**Tracy et al. 1983**
	0.017	**Sheppard et al. 1989**
	0.003[b] (0.0009 – 0.0088)	**Lakshmanan and Venkateswarlu 1988**
***Solidago* sp. (goldenrod)**	**0.017 (0.0056 – 0.043)**	**Sheppard et al. 2004**
***Sphagnum* spp. (sphagnum moss)**	**0.126**	**Sheppard and Thibault 1984**
***Spinacia oleracea* (spinach)**	**0.018**	**Sheppard et al. 1989**
***Symphoricarpos* sp. (snowberry)**	**0.025**	**Sheppard et al. 2004**
***Tanacetum vulgare* (common tansy)**	**0.038**	**Sheppard et al.2004**
***Thuja occidentalis* (cedar)**	**0.010**	**Sheppard et al. 2004**
***Triticum aestivum* (wheat)**	**0.164 (0.03 – 0.9)**	**Shahandeh and Hossner 2002**
***Typha angustifolia* (cattail)**	**0.006 (0.0036 – 0.0085)**	**Sheppard et al. 2004**
***Typha latifolia* (cattail)**	**0.014 (0.001 – 0.715)**	**Amiro and Sheppard 1987**
***Vaccinium* sp. (blueberry)**	**0.018**	**Sheppard et al. 1989**
	0.0004	**Morton et al. 2002**
***Viburnum opulus* (high-bush cranberry)**	**0.0037 (0.0004 – 0.0173)**	**Sheppard et al. 2004**
***Vicia cracca* (tufted vetch)**	**0.028**	**Sheppard et al. 2004**
***Vitis riparia* (wild grapes)**	**0.0006 (0.00014 – 0.0028)**	**Sheppard et al. 2004**
***Zea mays* (corn)**	**0.015**	**Tracy et al. 1983**
	0.0004	**Sheppard et al. 1989**
***Zizania aquatica* (wild rice)**	**0.0005**	**Sheppard et al. 1989**
Unspecified vegetation	**0.063 (0.05 – 0.08)**	**Miera et al. 1980**
Lichens / Fungi		
***Cladonia* spp. (fructicose lichen)**	**0.297 (0.184 – 0.472)**	**Sheppard and Thibault 1984**
***Umbilicaria muhlenbergii* (umbilicate lichen)**	**0.735 (0.667 – 0.810)**	**Sheppard and Thibault 1984**
Invertebrates		
***Lumbricus terrestris* (earthworm)**	**0.455 (0.082 – 2.38)**	**Sheppard & Evenden 1992a**
Mammals		
***Peromyscus maniculatus* (deer mouse)**	**(~ 0.0001 – 0.001)**	**Miera et al. 1980**
***Thomomys bottae* (valley pocket gopher)**	**(~ 0.0001 – 0.01)**	**Miera et al. 1980**

[a]Concentration ratios were measured as [U] in dry tissue divided by [U] in dry soil (unless otherwise specified).
[b]These concentration ratios were calculated as [U] in fresh tissue (rather than dry) divided by [U] in dry soil.

Table 9. Selected and Consulted Studies on Microbial Processes.

Organism/Effect	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Extraction method	Reference
Selected							
Respiration	LOEC	500	$UO_2(OH)_2 \cdot H_2O$	7.27	fine loam; 24.3 g/kg total organic C	Nominal	Meyer *et al.* 1998a
Decomposition	LOEC	25,000	$UO_2(OH)_2 \cdot H_2O$	7.27	fine loam; 24.3 g/kg total organic C	Nominal	Meyer *et al.* 1998a
Nitrogen Mineralization	NOEC	25,000	$UO_2(OH)_2 \cdot H_2O$	7.27	fine loam; 24.3 g/kg total organic C	Nominal	Meyer *et al.* 1998a
Consulted							
Phosphatase activity	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	7.5	loam; 24% clay; 2.2% organic content	Nominal	Sheppard *et al.* 1992
	NOEC LOEC	100 1,000	$UO_2(NO_3)_2 \cdot H_2O$	6.2	fine sand; 2% clay; 1% organic content	Nominal	Sheppard *et al.* 1992
	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	7.5	fine sandy loam; 18% clay; 18.4% organic content	Nominal	Sheppard *et al.* 1992
	NOEC LOEC	100 1,000	$UO_2(NO_3)_2 \cdot H_2O$	4.9	fine sand; 2% clay; 0.7% organic content	Nominal	Sheppard *et al.* 1992
	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	5.5	loamy sand; 6% clay; 3.5% organic content fine sandy loam; 15% clay; 2.6% organic content	Nominal	Sheppard *et al.* 1992
	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	7.3	loamy fine sand; 4% clay; 0.8% organic content	Nominal	Sheppard *et al.* 1992
	NOEC LOEC	100 1,000	$UO_2(NO_3)_2 \cdot H_2O$	7.8	fine sandy loam; 13% clay; 5.7% organic content	Nominal	Sheppard *et al.* 1992
	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	6.6	clay loam; 33% clay; 3.1% organic content	Nominal	Sheppard *et al.* 1992
	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	7.0	fine sandy loam; 12% clay; 4.2% organic content	Nominal	Sheppard *et al.* 1992
	NOEC	>1,000	$UO_2(NO_3)_2 \cdot H_2O$	7.8		Nominal	Sheppard *et al.* 1992

Table 9. Selected and Consulted Studies on Microbial Processes Cont'd.

Organism/Effect Consulted	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Extraction method	Reference
Ferrous iron oxidation	NOEC LOEC	0.2 mM 0.4 mM	$UO_2SO_4 \cdot 3½H_2O$	1.3	Aqueous growth medium	Nominal	Tuovinen and Kelly 1974
Growth of *Zoogloea ramigera*	IC100 LOEC	1000 mg/L 1 mg/L	$UO_2(NO_3)_2 \cdot 6H_2O$	NR	Aqueous growth medium	Nominal	Norberg and Molin 1983

Table 10. Selected and Consulted Plant Toxicological Studies.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method*	Reference
Selected								
Alfalfa (*Medicago sativa*)	Shoot yield	NOEC	>100	$UO_2(NO_3)_2{\cdot}H_2O$	8.0 (loam) 8.6 (sand)	Loam; sand	Nominal	Sheppard *et al.* 1984
Swiss chard (*Beta vulgaris cicla*)	Shoot yield	NOEC	>100	$UO_2(NO_3)_2{\cdot}H_2O$	8.0 (loam) 8.6 (sand)	Loam; sand	Nominal	Sheppard *et al.* 1984
Scots pine (*Pinus sylvestris*)	Seedling mortality	NOEC	>100	$UO_2(NO_3)_2{\cdot}H_2O$	7.8-8.0	Clay loam	Nominal	Sheppard *et al.* 1985
Blueberry (*Vaccinium angustifolium*)	Mortality	LOEC NOEC	10,000 8,000	$UO_2(NO_3)_2{\cdot}H_2O$	4.8	Sphagnum peat	Nominal	Sheppard and Evenden 1988a
Beans (*Phaseolus vulgaris* cv Contender)	Seedlings emerged Seedling weight Seedling weight Pods at first pick First pod yield Final pod weight Final pod weight Final veg. weight Final veg. weight	NOEC NOEC LOEC NOEC NOEC NOEC LOEC NOEC LOEC	>1,000 300 1,000 >1,000 >1,000 300 1,000 300 1,000	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Seedlings emerged Seedling weight Pods at first pick First pod yield Final pod weight Final veg. weight	NOEC NOEC NOEC NOEC NOEC NOEC	>1,000 >1,000 >1,000 >1,000 >1,000 >1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	loam; 2.2% organic content	Nominal	Sheppard *et al.* 1992
	Seedlings emerged Seedlings emerged Seedling weight	NOEC LOEC NOEC	300 1,000 >1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992
Corn (*Zea mays*)	Germination Germination Seedling dry weight Seedling dry weight	NOEC LOEC NOEC LOEC	300 1,000 300 1,000	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Germination Seedling dry weight	NOEC NOEC	>1,000 >1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992

Table 10. Selected and Consulted Plant Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method*	Reference
Selected								
Lettuce (*Lactuca sativa*)	Germination	NOEC	300	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Germination	LOEC	1,000					
	Germination	NOEC	>1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992
Tomato (*Lycopersicon esculentum*)	Germination	NOEC	300	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Germination	LOEC	1,000					
	Germination	NOEC	>1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992
Brassica rapa	Germination	NOEC	300	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Germination	LOEC	1,000					
	Stem length	NOEC	3,000					
	Stem length	LOEC	10,000					
	Straw weight	NOEC	3,000					
	Straw weight	LOEC	10,000					
	Germination	NOEC	>10,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992
	Stem length	NOEC	3,000					
	Stem length	LOEC	10,000					
	Straw weight	NOEC	3,000					
	Straw weight	LOEC	10,000					
	Seed weight	NOEC	3,000					
	Seed weight	LOEC	10,000					
	Number of seeds	NOEC	3,000					
	Number of seeds	LOEC	10,000					
	Germ% of best seeds	NOEC	3,000					
	Germ% of best seeds							
	Overall %germination	LOEC	10,000					
	Overall %germinaton							
		NOEC	3,000					
		LOEC	10,000					

Table 10. Selected and Consulted Plant Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method*	Reference
Selected								
White pine (*Pinus strobus*)	Germination	NOEC	300	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Germination	LOEC	1,000					
	Rate of germination	NOEC	300					
	Rate of germination	LOEC	1,000					
	Germination	NOEC	>1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992
	Rate of germination	NOEC	>1,000					
Buffalograss (*Buchloe dactyloides*)	Percent emergence	NOEC	>25,000	$UO_2(OH)_2{\cdot}nH_20$	8.6	sand supplemented with Osmocote 120-d slow release 14-14-14 fertilizer	Nominal	Meyer and McLendon 1997
	Seedling survival	NOEC	>25,000					
	Plant survival	NOEC	5,000					
	Plant survival	LOEC	25,000					
	Shoot biomass	NOEC	5,000					
	Shoot biomass	LOEC	25,000					
	No. inflorescences	NOEC	5,000					
	No. inflorescences	LOEC	25,000					
	Inflorescence weight	NOEC	5,000					
	Inflorescence weight							
	Stolon Production	LOEC	25,000					
	Stolon Production							
		NOEC	5,000					
		LOEC	25,000					
Little bluestem (*Schizachyrium scoparium*)	Percent emergence	NOEC	>25,000	$UO_2(OH)_2{\cdot}nH_20$	8.6	sand supplemented with Osmocote 120-d slow release 14-14-14 fertilizer	Nominal	Meyer and McLendon 1997
	Seedling survival	NOEC	>25,000					
	Plant survival	NOEC	5,000					
	Plant survival	LOEC	25,000					
	Shoot biomass	NOEC	5,000					
	Shoot biomass	LOEC	25,000					

Table 10. Selected and Consulted Plant Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method*	Reference
Selected								
Purple threeawn (*Aristida purpurea*)	Percent emergence	NOEC	>25,000	$UO_2(OH)_2 \cdot nH_20$	8.6	sand supplemented with Osmocote 120-d slow release 14-14-14 fertilizer	Nominal	Meyer and McLendon 1997
	Seedling survival	NOEC	>25,000					
	Plant survival	NOEC	5,000					
	Plant survival	LOEC	25,000					
	Shoot biomass	NOEC	5,000					
	Shoot biomass	LOEC	25,000					
	No. inflorescences	NOEC	5,000					
	No. inflorescences	LOEC	25,000					
	Inflorescence weight	NOEC	5,000					
	Inflorescence weight	LOEC	25,000					
Northern wheatgrass (*Elymus lanceolatus*)	Emergence	NOEC	>3190	Uranyl nitrate	7.5	Loam, 2.2% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Shoot length	NOEC	838					
	Shoot length	LOEC	3190					
	Root length	NOEC	838					
	Root length	LOEC	3190					
	Plant dry biomass	NOEC	838					
	Plant dry biomass	LOEC	3190					
	Emergence	NOEC	>2580	Uranyl nitrate	6.2	Limed fine sand, 1.0% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Shoot length	NOEC	>2580					
	Root length	NOEC	994					
	Root length	LOEC	2580					
	Plant dry biomass	NOEC	>2580				Nominal	Gulati et al. 1980
Wheat (*Triticum* sp.?)	Yield	NOEC	>6	Uranyl acetate	8.5	Sandy-loam; 0.09% organic carbon		
						Sandy-loam; 0.09% organic carbon	Nominal	Gulati et al. 1980
Tomato § (*Lycopersicon esculentum*)	Yield	NOEC	1.5	Uranyl acetate	8.5			
		LOEC	3.0					

Table 10. Selected and Consulted Plant Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method*	Reference
Consulted								
Mung bean § (*Vigna radiata*)	Germination Germination Seedling vigour Seedling vigour Dry weight	NOEC LOEC NOEC LOEC LOEC	20 µg/mL 40 µg/mL 2.5 µg/mL 5 µg/mL 2.5 µg/mL	$UO_2(NO_3)_2 \cdot 7H_2O$ (uranyl nitrate)	NR	Filter paper and liquid growth medium	Nominal	Aery and Jain 1995
Wheat (*Triticum aestivum*)	Root growth Chlorophyll content	EC25 EC25	1.25 µg/mL 10 µg/mL	$UO_2(NO_3)_2 \cdot 7H_2O$ (uranyl nitrate)	NR	Filter paper and liquid growth medium	Nominal	Aery and Jain 1997
Wheat § (*Triticum aestivum*)	Shoot length Shoot length Root length Root length Shoot fresh biomass Shoot fresh biomass Root fresh biomass Root fresh biomass Shoot dry biomass Shoot dry biomass Root dry biomass Root dry biomass Leaf area Leaf area Spike number/plant Spike number/plant Seed number/spike Seed number/spike Seed weight Seed weight	NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC NOEC LOEC	5 25 5 25 5 25 1 5 1 5 1 5 5 25 5 25 0 1 1 5	N_2O_8U	7.8	Sandy loam: 2% organic content	Nominal	Aery and Jain 1998

Table 10. Selected and Consulted Plant Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method*	Reference
Consulted								
Sorghum (*Sorghum bicolor*?)	Yield	EC51	100	NR	NR	Non-saline soil	Nominal	Shevchenkov et al. 1987
Maize (*Zea mays*)	Yield	EC63	100	NR	NR	Non-saline soil	Nominal	Shevchenkov et al. 1987
Sugarbeet (*Beta vulgaris*)	Yield	EC25	100	NR	NR	Non-saline soil	Nominal	Shevchenkov et al. 1987
Cotton (*Gossypium* sp.?)	Yield	EC25	100	NR	NR	Non-saline soil	Nominal	Shevchenkov et al. 1987
Wheat (*Triticum* sp.?)	Yield	EC11	50	Uranyl nitrate	7.5	28% clay	Nominal	Zhukov and Xudilkin 1971
Soybean (*Glycine max*)	Germination Seedling length Seedling length Chlorophyll content Chlorophyll content	NOEC NOEC LOEC NOEC LOEC	>42 µg/mL 0.42 µg/mL 42 µg/mL 0.42 µg/mL 42 µg/mL	Uranium oxide	5 – 6.5	Hoagland's nutrient solution	Nominal	Murthy et al. 1984

NR = not reported

* Analytical method: ICP-MS = inductively coupled plasma-mass spectroscopy.

§ NOECs and LOECs were not statistically derived by the authors. NOECs were determined to be the highest concentration at which there was less than a 15% effect observed as compared to the controls. LOECs were determined to be the lowest concentration at which a 15% or greater effect was observed as compared to the controls.

Table 11. Selected and consulted invertebrate toxicological studies.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method	Reference
Selected								
Earthworm (*Lumbricus terrestris*)	Survival at 14 days	NOEC	>1,000	$UO_2(NO_3)_2{\cdot}H_2O$	6.2	Limed fine sand; 1% organic content	Nominal	Sheppard *et al.* 1992
	Survival at 75 days	NOEC	300					
	Survival at 75 days	LOEC	1,000					
	Fresh weight	NOEC	>300					
	Dry weight	NOEC	>300					
	Survival at 14 days	NOEC	>1,000	$UO_2(NO_3)_2{\cdot}H_2O$	7.5	Fine sandy loam; 18.4% organic content	Nominal	Sheppard *et al.* 1992
	Survival at 75 days	NOEC	>1,000					
	Fresh weight	NOEC	>1,000					
	Dry weight	NOEC	>1,000					

Table 11. Selected and Consulted Invertebrate Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method	Reference
Selected								
Earthworm (*Eisenia andrei*)	Survival at 14 days	NOEC	>838	Uranyl nitrate	7.5	Loam; 2.2% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	NOEC	>838					
	Number of hatched cocoons	NOEC	>838					
	Number of unhatched cocoons	NOEC	>838					
	Juvenile wet mass	NOEC	>838					
	Survival at 14 days	NOEC	>994	Uranyl nitrate	7.5	Fine sandy loam; 18.4% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	NOEC	>994					
	Number of hatched cocoons	NOEC	>994					
	Number of unhatched cocoons	NOEC	>994					
	Juvenile wet mass	NOEC	>994					
	Survival at 14 days	NOEC	>1120	Uranyl nitrate	6.2	Limed fine sand; 1% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	NOEC	>1120					
	Number of hatched cocoons	NOEC	>1120					
	Number of unhatched cocoons	NOEC	>1120					
	Juvenile wet mass	NOEC	>1120					
	Adult survival	NOEC	838	Uranyl nitrate	7.5	Loam; 2.2% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Adult survival	LOEC	3190					
	Number of juveniles	NOEC	838					
	Number of juveniles	LOEC	3190					
	Fecundity	NOEC	838					
	Fedundity	LOEC	3190					

Table 11. Selected and Consulted Invertebrate Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method	Reference
Selected								
Collembolan / Springtail (*Onychiurus folsomi*)	Adult survival	NOEC	1	Uranyl nitrate	6.2	Limed fine sand; 1% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Adult survival	LOEC	994					
	Number of juveniles	NOEC	994					
	Number of juveniles	LOEC	2580					
	Fecundity	NOEC	1					
	Fecundity	LOEC	994					
Collembolan / Springtail (*Onychiurus folsomi*)	Adult survival	EC20	390	Uranyl nitrate	7.5	Loam; 2.2% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	EC20	910					
	Adult survival	EC20	480	Uranyl nitrate	7.5	Fine sandy loam; 18.4% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	EC20	1030					
	Adult survival	EC20	190, 92	Uranyl nitrate	6.2	Limed fine sand; 1% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	EC20	150					
Collembolan / Springtail (*Folsomia candida*)	Adult survival	EC20	710	Uranyl nitrate	7.5	Loam; 2.2% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	EC20	840					
	Adult survival	EC20	1030	Uranyl nitrate	7.5	Fine sandy loam; 18.4% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	EC20	1900					
	Adult survival	EC20	>1000[a], 350[a]	Uranyl nitrate	7.5	Limed fine sand; 1% organic matter	Aqua regia digestion and ICP-MS	Sheppard et al. 2004
	Number of juveniles	EC20	2200[a], >3000[a]					

Table 11. Selected and Consulted Invertebrate Toxicological Studies Cont'd.

Organism	Effect (% decrease)	Endpoint	Concentration (mg/kg)	Form of uranium	Soil pH	Test Substrate	Analytical method	Reference
Consulted								
Earthworm (*Eisenia fetida andrei*)	Survival at 96 hours	LC50	40 µg/cm^2	Uranium acetate	NR	Moistened filter paper in a petri dish	Nominal	Ribera et al. 1996
Earthworm (*Eisenia fetida andrei*)	Survival at 96 hours	LC50	13.5 µg/cm^2	Uranyl acetate	NR	Moistened filter paper in a petri dish	Nominal	Labrot et al. 1999
Earthworm (*Eisenia fetida andrei*)	Reduced acetyl-cholinesterase	EC32	0.5 µg/cm^2	Uranyl acetate	NR	Moistened filter paper in a petri dish	Nominal	Labrot et al. 1996
	Reduced malondialdehyde	EC27	1.0 µg/cm^2					

[a] not significant
ICP-MS = inductively coupled plasma-mass spectroscopy

Table 12. Toxicity of Uranium to Livestock and Terrestrial Wildlife.

Organism	Body Weight (kg)	Ingestion Rate (g/d)	Effect	Endpoint	Exposure Concentrations (mg/kg bw/)	Source of uranium	Exposure period	Exposure route	Estimated dose (mg/kg bw/d)	Reference
Leghorn cockeral chicks	NA	NA	Mortality Mortality	LC_{50} LOEC	>235 160	N_2O_8U	7 days	Subcutaneous injection, single dose	NA	Harvey *et al.* 1986
American black ducks (*Anas rubripes*)	1.25	125†	Mortality Body weight Liver weights Kidney weights Gross lesions on organs	NOEC	0, 2.5, 10, 40, 160	powdered U	6 weeks	Oral	>160	Haseltine and Sileo 1983
Japanese quail (*Coturnix coturnix japonica*)	0.108	8.4‡	Kidney lesions	LOEC	0, 0.04, 12	N_2O_8U	18 hours	Injection into pectoral muscles, single dose	0.04*	Kupsh *et al.* 1991
Dairy cattle	650**	16,900††	General health Milk yield	LOEC	0.615	$UO_2(NO_3)_2 \cdot H_20$	2 weeks	Oral	0.615	Garner, 1963

* Units are mg/kg bw for the single dose study.
† Food ingestion rate was not specified in original study and is therefore based on that derived by Sample *et al.* (1996) for the same organism.
‡ Food ingestion rate was not specified in original study and is therefore based on the food ingestion rate of the bobwhite quail (U.S. EPA 1993).
** Estimated by Puls (1994).
†† Assuming cattle ingest about 2.6% of their body weight on a daily basis (Puls 1994).

Table 13. Toxicity of Uranium to Other Mammalian Species.

Organism	Body Weight‡ (kg)	Ingestion Rate‡ (g/d)	Effect‡	Endpoint	Estimated dose (mg/kg bw/d)	Exposure Concentrations (mg/kg bw/d)‡	Source of uranium	Exposure period	Exposure route	Reference
Swiss mouse (M)	0.025-0.03	NA¶	Mortality	LC_{50}	148	27, 49, 88, 159, 286, 514	$UO_2(C_2H_3O_2)_2$	single dose	gavage	Domingo *et al.* 1987
Sprague-Dawley rat (M)	0.25-0.3	23†	Mortality Renal/Hepatic	LC_{50} LOEL	125, 130	12, 25, 49, 98, 130, 196, 392, 785	$UO_2(C_2H_3O_2)_2$	single dose	gavage	Domingo *et al.* 1987
Swiss mouse	0.026-0.03	NA**	Embryolethality Embryolethality Reproductive Reproductive	NOEL LOEL NOEL LOEL	6, 14, 6, 14	0, 3, 6, 14	$UO_2(C_2H_3O_2)_2$	60 d (male); 14 d before pregnancy to weaning (female)	oral	Paternain *et al.* 1989
Swiss mouse (female)	0.025-0.03	NA**	Fetotoxicity Teratogenicity Embryolethality Maternal toxicity	LOEL LOEL NOEL LOEL	<2.8 <2.8 >28 <2.8	0, 2.8, 5.6, 14, 28	$UO_2(C_2H_3O_2)_2$	Gestation days 6-15	gavage	Domingo *et al.* 1989a
Swiss mouse (female)	0.026-0.03	NA**	Reproductive Reproductive	NOEL LOEL	3, 30	0, 0.03, 0.3, 3, 30	$UO_2(C_2H_3O_2)_2$	Day 13 of pregnancy to day 21 post-birth	gavage	Domingo *et al.* 1989b
Rabbit	2.86-3.1††	94-100†	Renal Growth Growth	LOEL NOEL LOEL	2.8, 2.8, 14	0, 2.8, 14, 71	$UO_2(NO_3)_2{\cdot}6H_2O$	30 d	oral	Maynard and Hodge 1949
Rat	not specified by U.S. EPA (1998)	not specified by U.S. EPA (1998)	Renal Renal Renal Renal Renal Mortality Mortality	NOEL LOEL LOEL LOEL LOEL LC_{50} LC_{50}	153, 163, 39, 120, 160 591, 386	not specified by U.S. EPA (1998)	$UO_2(C_2H_3O_2)_2$ $UO_2(C_2H_3O_2)_2$ UO_2F_2 $UO_2(NO_3)_2$ UCl_4 UO_4 UO_2F_2	30 d	oral	Maynard and Hodge 1949
Dog	10.1-10.8††	200-210†	Renal	LOEL LOEL LOEL	7.7 9.5 132	not specified by U.S. EPA (1998)	UO_2F_2 $UO_2(NO_3)_2$ UCl_4	30 d	oral	Maynard and Hodge 1949

Table 13. Toxicity of Uranium to Other Mammalian Species Cont'd.

Organism	Body Weight‡ (kg)	Ingestion Rate‡ (g/d)	Effect‡	Endpoint	Estimated dose (mg/kg bw/d)	Exposure Concentrations (mg/kg bw/d)‡	Source of uranium	Exposure period	Exposure route	Reference
Rat	not specified by U.S. EPA (1998)	not specified by U.S. EPA (1998)	Reproductive Mortality Renal Hematological	LOEL LOEL LOEL LOEL	470, 470, 237, 237	not specified by U.S. EPA (1998)	$UO_2(NO_3)_2$	2 y 2 y 2 y 2 y	oral	Maynard and Hodge 1949
Rat	not specified by U.S. EPA (1998)	not specified by U.S. EPA (1998)	Renal Hematological Growth Growth	NOEL NOEL NOEL LOEL	24, 118, 12, 24	not specified by U.S. EPA (1998)	$UO_2(NO_3)_2$	1 y 1 y 2 y 2 y	oral	Maynard *et al.* 1953
Sprague-Dawley rat (male)	0.07-0.09	23†	Mortality Hematological Hematological Hepatic/renal Hepatic/renal	NOEL NOEL LOEL NOEL LOEL	1, 4.5, 9, 4.5, 9	0, 1, 2, 4.5, 9	$UO_2(C_2H_3O_2)_2$	4 wk	drinking water	Ortega *et al.* 1989
New Zealand white rabbit (not pathogen-free)	4.0 (M)	151 (M)	Renal (M)	LOEL	0.05	0, 0.05, 0.2, 0.88, 4.82, 28.7 (M)	$UO_2(NO_3)_2{\cdot}6H_2O$	91 d	drinking water	Gilman *et al.* 1998a
	3.9 (F)	228 (F)	Renal (F)	LOEL	0.49	0, 0.49, 1.32, 43.02 (F)	$UO_2(NO_3)_2{\cdot}6H_2O$	91 d	drinking water	Gilman *et al.* 1998a
New Zealand white rabbit (pathogen-free)	3.5 (M)	211 (M)	Renal (M)	LOEL	1.36	0, 1.36, 40.96 (M)	$UO_2(NO_3)_2{\cdot}6H_2O$	91 d	drinking water	Gilman *et al.* 1998b
Sprague-Dawley rat (female)	0.309-0.337	19.56-20.44	Renal	LOAEL	0.09	<0.0001, 0.09, 0.42, 2.01, 9.98, 53.56	$UO_2(NO_3)_2{\cdot}6H_2O$	91 d	drinking water	Gilman *et al.* 1998c
Sprague-Dawley rat (male)	0.487-0.522	21.23-24.14	Renal	LOAEL	0.06	<0.0001, 0.06, 0.31, 1.52, 7.54, 36.73	$UO_2(NO_3)_2{\cdot}6H_2O$	91 d	drinking water	Gilman *et al.* 1998c

Table 13. Toxicity of Uranium to Other Mammalian Species Cont'd.

Organism	Body Weight‡ (kg)	Ingestion Rate‡ (g/d)	Effect‡	Endpoint	Estimated dose (mg/kg bw/d)	Exposure Concentrations (mg/kg bw/d)‡	Source of uranium	Exposure period	Exposure route	Reference
Swiss mouse (M)	0.038	6.81-7.98 ml/L	No. pregnant F No. implantation No. early resorp. No. late resorp. No. dead fetus No. live fetus Body weight Body weight Testis weight Epididymis wt. Epididymis wt. Sperm count %motile cells %deform sperm	LOEL NOEL NOEL NOEL NOEL NOEL NOEL LOEL NOEL NOEL LOEL LOEL NOEL NOEL	<10 >80 >80 >80 >80 >80 40 80 >80 10 20 <10 >80 >80	0, 10, 20, 40, 80	$UO_2(C_2H_3O_2)_2$	64 d	drinking water	Llobet *et al.* 1991
Rat	not specified by ATSDR (1999)	not specified by ATSDR (1999)	Mortality Renal Renal	LC_{50} NOEC LOEC	12,000* 1360* 580*	not specified by ATSDR (1999)	UF_6	10 min 2 min 5 min	inhalation	Leach *et al.* 1984
Guinea Pig	0.39-0.48††	30-32†	Mortality	LC_{50}	62,000*	not specified by ATSDR (1999)	UF_6	2 min	inhalation	Leach *et al.* 1984
Rabbit	2.86-3.1††	94-100†	Mortality Renal Growth Growth	LC_{50} LOEC NOEC LOEC	0.25* 0.2* 0.2* 2*	0.04-10*	$UO_2(NO_3)_2 \cdot 6H_2O$ various various various	6.5 mth 7.5 mth 6.5 mth 6.5 mth	inhalation	Stokinger *et al.* 1953
Rat	not specified by ATSDR (1999)	not specified by ATSDR (1999)	Renal Renal Hematological	NOEC LOEC NOEC	0.05* 0.05* 10*	0.04-10*	various	1 y	inhalation	Stokinger *et al.* 1953
Dog	10.1-10.8††	200-210†	Respiratory Hematological	LOEC NOEC	5.1* 5.1*	5.1*	UF_6	1-5 y	inhalation	Leach *et al.* 1970, 1973
Monkey	8-10.9††	330-430†	Respiratory	LOEC	5.1*	5.1*	UF_6	1-5 y	inhalation	Leach *et al.* 1970, 1973
Guinea pig	0.39-0.48††	30-32†	Renal	NOEC LOEC	0.05* 0.2*	0.04-10*	various	9 mth 7.5 mth	inhalation	Stokinger *et al.* 1953

Table 13. Toxicity of Uranium to Other Mammalian Species Cont'd.

Organism	Body Weight‡ (kg)	Ingestion Rate‡ (g/d)	Effect‡	Endpoint	Estimated dose (mg/kg bw/d)	Exposure Concentrations (mg/kg bw/d)‡	Source of uranium	Exposure period	Exposure route	Reference
Beagle dog (female)	7.2	74†	Renal	LOEL	0.01	0.11-1.46§	$UO_2F_2 \cdot 2H_2O$	0.5-2.5 h	inhalation	Morrow *et al.* 1982a
Wistar rats (female)	0.16	16†	Mortality	NOEL LOEL	8.8 44	8.8, 44, 440, 880	UO_2	single dose	sub-cutaneous implantation	de Rey *et al.* 1984
Wistar rats, albino (female)	0.156††	16†	Renal	LOEL	<0.05	0.05-0.5	$UO_2(NO_3)_2$	single dose	parenteral injection	Bentley *et al.* 1985
Wistar rat (male)	0.354	35†	Renal	NOEL LOEL	0.04 12	0, 0.04, 12	$UO_2(NO_3)_2$	single dose	injection in femoral vein	Kupsh *et al.* 1991
Rabbit	3	94-100†	Mortality	NOEL LOEL	0.42 0.54	0.3, 0.4, 0.5, 0.7, 0.8, 1.1, 3.4	$UO_2(NO_3)_2$	single dose	intravenous injection	Gawlik *et al.* 1976
Rabbit	2.86-3.1††	94-100†	Mortality Weight loss Weight loss Skin irritation Renal	LD_{50} NOEL LOEL LOEL LOEL	28 6 30 1.4 1.4	not specified in ATSDR (1999)	$UO_2(NO_3)_2 \cdot 6H_2O$	4 h	dermal	Orcutt 1949
Rabbit	2.86-3.1††	94-100†	Renal/weight loss Dermal Ulcers	LOEL LOEL	2.3 2.3	not specified in ATSDR (1999)	$UO_2(NO_3)_2 \cdot 6H_2O$	5 wk	dermal	Orcutt 1949
Rat	not specified in ATSDR (1990)	not specified in ATSDR (1999)	Mortality Renal/weight loss	LD_{50} LOEL	101 85	not specified in ATSDR (1999)	$UO_2(NO_3)_2 \cdot 6H_2O$	4 h	dermal	Orcutt 1949
Guinea pig	0.39-0.48††	30-32†	Mortality Renal/dermal Weight loss Weight loss	LD_{50} LOEL NOEL LOEL	1190 47 47 161	not specified in ATSDR (1999)	$UO_2(NO_3)_2 \cdot 6H_2O$	4 h 4 wk 4 wk 4 wk	dermal	Orcutt 1949
Mouse	not specified in ATSDR (1999)	not specified in ATSDR (1999)	Mortality Renal	LD_{50} LOEL	7600 948	not specified in ATSDR (1999)	$UO_2(NO_3)_2 \cdot 6H_2O$	4 h	dermal	Orcutt 1949

Table 13. Toxicity of Uranium to Other Mammalian Species Cont'd.

Organism	Body Weight‡ (kg)	Ingestion Rate‡ (g/d)	Effect‡	Endpoint	Estimated dose (mg/kg bw/d)	Exposure Concentrations (mg/kg bw/d)‡	Source of uranium	Exposure period	Exposure route	Reference

* Units are mg U/ m^3

† Food ingestion rate was not specified in original study and is therefore based on that determined by the U.S. EPA (1988) for the same organism.

‡ M=male; F=female.

§ Units are mg U/kg.

¶ Food ingestion rate was not specified in original study and currently there are no published estimates of the food ingestion rates of Swiss mice; however, route of exposure was a single dose, and food ingestion rate was not required to calculate the dose.

** Food ingestion rate was not specified in original study and currently there are no published estimates of the food ingestion rates of Swiss mice; however, food ingestion rate was not required to calculate dose.

†† Body weight was not specified in original study and is therefore based on that determined by the U.S. EPA (1988) for the same organism.

Table 14. Typical Values for Average Body Weights and Intakes of Air, Water and Soil by the Canadian General Population

Age (years)	Body weight[1] (kg)	Air intake[1] (m^3/d)	Water intake[1] (L/d)	Soil intake[2] (g/d)
0-6 months	8.2	2.1	0.75 (water mixed with formula or breast milk)[3]	0.02
7 months-4	16.5	9.3	0.6	0.08
5-11	32.9	14.5	0.8	0.02
12-19	59.7	15.8	1.0	0.02
20+	70.7	15.8	1.5	0.02

[1]Richardson 1997

[2]CCME 2005

[3]Health Canada 2004

Table 15. Mean Consumption Rate of Various Food Groups by Canadians, From 1972 Nutrition Canada Survey

FOOD GROUP	Consumption (g/person/day) 7 mo. - 4 yrs	5-11 yrs	12 – 19 yrs	20 + yrs
BABY FOODS	3.31	0.20	0.00	0.06
MILK & DAIRY	578.75	591.02	545.52	264.73
MEAT, POULTRY & EGGS	160.91	205.21	242.72	212.01
FISH & SHELLFISH	5.07	9.59	12.11	17.31
ROOT VEGETABLES	79.02	132.35	185.17	153.66
OTHER VEGETABLES	81.84	141.94	277.57	842.20
FRUITS & JUICES	179.76	190.89	155.83	163.26
CEREALS & GRAINS	165.63	266.72	282.41	220.56
SUGAR & SWEETS	161.89	257.75	328.37	168.31
FATS, NUTS & OILS	18.04	32.32	40.81	33.55
ALCOHOLIC DRINKS	0.59	2.11	23.50	104.96

Source: M. Richardson, Health Canada, pers comm. 2005

Table 16. Estimated Daily Uranium Intake via Food Consumption for the Canadian General Population

Food Group	Mean[1] (mg/kg fresh weight)	Daily mean intake from consumption of food[2] (µg U/person/day)			
		7 mo-4 yrs[2]	5-11 yrs	12-19 yrs	20 yrs+
BABY FOODS	0.00002	6.6E-05	4.0E-06	0.0E+00	1.2E-06
MILK & DAIRY	0.00002 to 0.0016	4.90E-02	6.64E-02	6.92E-02	4.95E-02
MEAT, POULTRY & EGGS	0.00003 to 0.0023	1.78E-01	1.97E-01	2.12E-01	1.97E-01
FISH & SHELLFISH	0.0035	1.77E-02	3.36E-02	4.24E-02	6.06E-02
ROOT VEGETABLES	0.0005 to 0.0012	3.98E-02	6.70E-02	9.34E-02	7.80E-02
OTHER VEGETABLES	0.00004 to 0.0012	5.80E-02	8.56E-02	9.72E-02	1.24E-01
FRUITS & JUICES	0.0001 to 0.0012	4.47E-02	3.85E-02	3.38E-02	4.19E-02
CEREALS & GRAINS	0.0018 to 0.0035	3.76E-01	6.45E-01	7.14E-01	5.66E-01
SUGAR & SWEETS	0.0012	1.94E-01	3.09E-01	3.94E-01	2.02E-01
FATS, NUTS & OILS	0.0006 to 0.0007	1.11E-02	2.01E-02	2.52E-02	2.04E-02
ALCOHOLIC DRINKS	0.00004	2.37E-05	8.45E-05	9.40E-04	4.20E-03
	Total (µg/day)	1.0	1.5	1.7	1.3

[1]Concentrations for ^{238}U in 2001 UK Total Diet Study in Table 15.

[2]No dietary intakes were estimated for infants as no data were identified for uranium in either breast milk or formula**.**

Table 17. Estimated Total Daily Uranium Intake by Age Class for the Canadian General Population[1]

Medium	Typical Uranium Levels	Daily Uranium Intake in µg/person/day				
		0 – 6 mo.	7 mo. - 4 yrs	5-11 yrs	12 - 19 yrs	20 + yrs
Air[2]	0.0001 µg/m^3	0.0002	0.00093	0.00145	0.00158	0.00158
Drinking water[3]	0.2 µg/L	0.15	0.12	0.16	0.2	0.3
Soil[4]	2 µg/g	0.04	0.16	0.04	0.04	0.04
Food[5]	0.00002 to 0.0035 µg/g	-	1.0	1.5	1.7	1.3
	Total intake via all routes (µg/day)	0.19	1.3	1.7	1.9	1.6
	Total intake (µg/kg bw/day)	0.023	0.078	0.052	0.033	0.023

[1]Based on body weights and intake rates in Table 14.

[2]Based on Ontario background air concentration of 0.1 ng U/m^3.

[3]Based on 0.2 µg U/L in Ontario drinking water.

[4]Assuming a background soil concentration of 2 µg/g (reported for uranium in ON and NB (see Table 2).

[5]Based on food intake rates in Table 16.

Table 18. Soil Quality Guidelines and Check Values for Uranium (mg/kg).

	Land use			
	Agricultural	**Residential/ parkland**	**Commercial**	**Industrial**
Guideline	**23[a]**	**23[a]**	**33[a]**	**300[a]**
Human health guidelines/check values				
$PSQG_{HH}$	23[b]	23[b]	33[b]	300[b]
Direct contact guideline	23	23	33	510
Inhalation of indoor air check	NC[c]	NC[c]	NC[c]	NC[c]
Off-site migration check	—	—	—	300
Groundwater check (drinking water)	NC[d]	NC[d]	NC[d]	NC[d]
Produce, meat and milk check	NC[d]	NC[d]	—	—
Environmental health guidelines/check values				
SQG_E	33[e]	500[f]	2000[f]	2000[f]
Soil contact guideline	500	500	2000	2000
Soil and food ingestion guideline	33	—	—	—
Nutrient and energy cycling check	NC[g]	NC[g]	NC[g]	NC[g]
Off-site migration check	—	—	—	7100
Groundwater check (aquatic life)	NC[d]	NC[d]	NC[d]	NC[d]
Interim Soil Quality Criteria (CCME 1991)	no value	no value	no value	no value

Notes: NC = not calculated; ND = not determined; $PSQG_{HH}$ = preliminary soil quality guideline for human health; SQG_E = soil quality guideline for environmental health. A dash indicates a guideline/check value that is not part of the exposure scenario for that land use and therefore is not calculated.

[a]Data are sufficient and adequate to calculate a $PSQG_{HH}$ and an SQG_E for this land use. The lower of the $PSQG_{HH}$ or the SQG_E becomes the soil quality guideline for this land use.

[b]The SQG_{HH} is the lowest of the human health guidelines and check values.

[c]The inhalation of indoor air check applies to volatile organic compounds and is not calculated for metal contaminants.

[d]Applies to organic compounds and thus is not calculated for metal contaminants. Concerns about metal contaminants should be addressed on a site-specific basis.

[e]The SQG_E for this land use is the lower of the soil contact guideline and the soil and food ingestion guideline.

[f]The SQG_E for this land use is based on the soil contact guideline.

[g]Data are insufficient/inadequate to calculate this check value.