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Considerations in the Development of Generic Human Health-Based Soil Quality Guidelines for Uranium

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ABSTRACT

At the current time in Canada, no human health-based generic (national) soil quality guidelines (SQG_{HH}) exist for uranium. This paper explores some of the various methods that could be used to develop human health soil guideline values for this element. To determine possible values that may be appropriate for use in Canada, the authors have applied both the existing CCME approach (1996) and the draft revised CCME approach (2005) to estimate possible SQG_{HH} for uranium for various land uses.

I. INTRODUCTION

Generic human health-based soil quality guidelines (SQG_{HH}) are used by environmental scientists to determine whether sites are considered to be appreciably impacted or contaminated by organic and inorganic substances, and thereby require further detailed risk assessment or remediation. In Canada, generic SQG_{HH} are developed by the Canadian Council of Ministers of the Environment (CCME), Health Canada, and various provincial agencies.

CCME currently has no published soil quality guidelines for uranium. For the past 2 to 3 years, CCME has been formulating soil quality guidelines for this element, and a companion technical supporting document. However, in 2004/05, CCME adopted revised procedures for the derivation of generic, national human health-based soil quality guidelines (CCME, 2005), procedures developed with the input and support of Health Canada. This paper presents some of the issues that are being considered in the development of generic SQG_{HH} for uranium. In addition, the paper compares the SQG_{HH} that would be developed using the previous CCME (1996) approach versus the revised methodology (CCME, 2005).

It is stressed that the SQG_{HH} presented in this paper are not to be interpreted as Health Canada or CCME guidance, but instead represent the authors' attempt of applying available information to two different guidelines derivation approaches, and to demonstrate the SQG_{HH} development process as it relates to uranium. It remains quite possible that Health Canada and/or the CCME may adopt different values for uranium than considered in this paper.

It must also be noted that this paper addresses neither radiological issues nor ecological issues associated with uranium in impacted soils. Instead, the focus of this paper is the chemical toxicity of uranium and the associated soil concentrations that may be considered to be adequate to screen contaminated sites for potential human health concerns.

II. GUIDELINES DERIVATION PROCEDURES

The CCME (and Health Canada) approach to deriving human health-based soil quality guidelines has three unique considerations, compared to methods espoused by the USEPA and other international agencies. First, background daily intake (from food, air, water and other sources that are unrelated to contaminated soils) is deducted from the tolerable daily intake (TDI; the toxicological reference value considered to protect the vast majority of Canadians from non-carcinogenic health effects of a substance; analogous to the US EPA RfD) to determine the 'residual TDI' (RTDI). This is meant to ensure that tolerable 'contamination' in soil will not result in total exposure (contaminated site + background) exceeding the TDI.

Secondly, the background soil concentration (from natural sources for inorganic elements) is the starting place for guidelines development. It is considered inappropriate to establish a guideline that is lower than a concentration in soil considered to be natural in origin. It is recognized that natural levels of inorganic elements vary geographically across Canada, and the CCME guidelines development process recognizes the need to modify the background soil concentration (BSC) for local geological considerations.

Finally, it is recognized that environmental quality guidelines are also developed for air, water, food, and some consumer products, as well as soil. Therefore, to ensure that simultaneous exposure to all five media at their guideline limits will not result in total exposure exceeding the RTDI, one fifth (0.2) of the RTDI is committed to the derivation of a soil quality guideline.

In CCME (1996), the equation to estimate generic SQG_{HH} for threshold response chemicals (i.e., chemicals with toxicity reference values expressed as TDI) was:

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

where,

- SQG_{HH} = human health soil quality guideline (mg/kg)
- TDI = tolerable daily intake of the chemical (mg/kg bw/day)
- EDI = estimated daily intake of the chemical for a typical Canadian (mg/kg bw/day)
- SF = soil allocation factor of (20% or 0.2 by default)
- BW = body weight for receptor of concern (kg)
- SIR = soil ingestion rate for receptor of concern (kg/d)
- IR_S = soil inhalation rate (from particulates) for receptor of concern (kg/d)
- SR = soil dermal contact rate for receptor of concern (kg/d)
- BSC = background soil concentration (mg/kg)
- AF_G = relative absorption factor of chemical across the gastrointestinal tract
- AF_S = relative absorption factor of chemical across the skin
- AF_L = relative absorption factor of chemical across the respiratory tract
- ET = exposure term (fraction of time at the site) (unitless)

In the draft revised CCME approach (CCME, 2005), the equation for estimation of SQG_{HH} for threshold response chemicals changed very little, with the addition of a time-dependant factor in consideration of inhalation exposure:

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

where,

- ET_1 = exposure term 1 (fraction of days per year at the site) (unitless)
- ET_2 = exposure term 2 (fraction of hours per day at the site) (unitless)

The most significant revisions to the guidelines derivation process in the 2005 approach relate to the assumptions used to quantify the various input parameters within the equation. These are discussed below.

III. INPUT PARAMETERS AVAILABLE TO ESTIMATE GENERIC SQG_{HH}

Receptor Characteristics

Under both the original and revised CCME approaches, the critical receptor for agricultural, residential and commercial land uses was generally identified as a young child (generally defined as a ‘toddler’; 0.5 to 4 years of age). This age group has the greatest intake of air, water, soil and food per unit of body weight, compared to infants (0 to 0.5 years), children (5 to 11 years), teens (12 to 19 years) and adults (20+ years). Adults are deemed the common critical receptors for industrial land uses. The receptor characteristics for all age groups defined in 1996 and those recommended in 2005 are presented in Table 1. The revised characteristics reflect new data, and in many cases specific Canadian data, that were not available in 1996.

Table 1 Comparison of Various Receptor Parameters used in CCME (1996) versus CCME (2005) Soil Quality Guideline Approaches

Parameter	CCME (1996) Value	CCME (2005) Value
Body Weight	Toddler = 13 kg Adult = 70 kg	Toddler = 16.5 kg Adult = 70.7 kg
Soil Ingestion Rate	Toddler = 80 mg/day (or 8 x 10^{-5} kg/day) Adult = 20 mg/day (or 2 x 10^{-5} kg/day)	Toddler = 80 mg/day (or 8 x 10^{-5} kg/day) Adult = 20 mg/day (or 2 x 10^{-5} kg/day)
Inhalation Rate	Toddler = 5 m^3/day Adult = 23 m^3/day	Toddler = 9.3 m^3/day Adult = 15.8 m^3/day
Soil Particle Inhalation Rate	Toddler = 2.5 x 10^{-7} kg/day Adult = 1.2 x 10^{-6} kg/day	Toddler = 7.1 x 10^{-9} kg/day Adult = 1.2 x 10^{-8} kg/day
Soil Adherence Factor	1 x 10^{-2} kg/m^2/event (entire body)	1 x 10^{-3} kg/m^2/event (hands) 1 x 10^{-4} kg/m^2/event (rest of body)
Skin Surface Area for Contact with Soil	Toddler = 0.26 m^2 (hands, arms and legs) Adult = 0.43 m^2 (hands and arms)	Toddler = 0.043 m^2 (hands); 0.26 m^2 (rest of body) Adult = 0.089 m^2 (hands); 0.25 m^2 (rest of body [i.e., arms])
Soil Dermal Contact Rate	Toddler = 2.6 x 10^{-3} kg/day Adult = 4.3 x 10^{-3} kg/day	Toddler = 6.9 x 10^{-5} kg/day Adult = 1.1 x 10^{-4} kg/day

Tolerable Daily Intake for Uranium

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 US Environmental Protection Agency [US EPA]) and Minimum Risk Level (MRL; used by the Agency for Toxic Substances and Disease Registry [US ATSDR]).

In the case of uranium, Health Canada, ATSDR, US EPA and WHO all offered toxicity reference values (TRVs) that were considered in the development of the SQG_{HH} and these are discussed below (and summarized in Table 2). 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.

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).

US EPA (2005) 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. US 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). US 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 US EPA (2005) reports that their RfD has not been revised since 1989 and it is not entirely clear if the US 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 exposures for uranium. WHO (2001) reported a Tolerable Intake (TI) of 0.5 µg/kg bw/day for 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 (2001) 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 (2004) provides a TDI of 0.6 µg/kg bw/day for uranium that is based on an identical rationale as provided by Health Canada (1999).

It is noted that Health Canada (1999), WHO (2001; 2004), US EPA (2005) 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.

Overall, Health Canada considers its TDI for uranium to be appropriate for use in the development of a Canadian SQG_{HH}. In the cases of the ATSDR (1999) and US EPA (2005), these international health agencies developed toxicity reference values that were considerably less conservative than the Health Canada value while WHO (2001) 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 (2004) offers a TDI value that is similar to Health Canada (1999).

Table 2 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	Rat	60	Renal toxicity	100	0.6
ATSDR	Rabbit	50	Renal toxicity	30	2
US EPA	Rabbit	2,800	Decreased body weight, renal toxicity	1,000	3
WHO (2001)	Rabbit	50	Renal toxicity	100	0.5
WHO (2004)	Rat	60	Renal toxicity	100	0.6

Estimated Daily Intake

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 include foods, soils, air and water. No specific consumer products were identified as a source of additional background U 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 (see Table 3).

The typical uranium airborne concentration used to estimate background exposure from inhalation was 1×10^{-4} μg/m^3. This value represents background atmospheric uranium concentrations in southern Ontario (Tracy and Prantl, 1985), the only Canadian data identified in this review. This air concentration is at the upper limit of the background uranium concentration range of 2.5×10^{-5} to 1×10^{-4} μg/m^3 reported by NCRP (1999) and higher than the mean concentration of 7.6×10^{-5} (reported in 1985/1986) for New York City (Fisenne et al., 1987).

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. com.). Slightly higher uranium concentrations were reported for drinking water in Quebec (0.35 to 0.97 μg/L) (Choiniere and Beaumier, 1997), and the Yukon (up to 7.2 μg/L) (E. Bergsam, Yukon Department of Environmental Health, pers. com.). 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 μg/g 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, 1999; 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. Guidelines are only screening tools, after all.

The total daily uranium intake via food was calculated using food intake rates for various age groups of Canadians (as per Richardson, 1997) and mean concentrations for naturally occurring uranium in food samples collected in New York City (Fisenne et al., 1987). No Canadian food concentration data was available in the literature reviewed. However, given the integration of the commercial food distribution system in North America, it was considered reasonable to assume that US data was applicable to the Canadian situation.

Consumption of cereal, vegetables, and meat, poultry, fish, and eggs contributed the greatest to uranium exposure (in decreasing order) for all age groups.

The EDI's provided in Table 3 were used in the derivation of the SQG_{HH} for both CCME (1996) and revised draft CCME methods.

Table 3 Estimated Daily Intake of Uranium for the Canadian General Population

Environmental Medium	**Assumed Uranium Concentrations in Environmental Media**	**Estimated Daily Intake (μg/kg bw/day)**	
		Young Child (0.5 – 4 yrs)	**Adult (>20 yrs)**
Air	1 x 10^{-4} μg/m^3	5.6 x 10^{-5}	2.2 x 10^{-5}
Drinking Water	0.2 μg/L	7.3 x 10^{-3}	4.2 x 10^{-3}
Soil	2 μg/g	9.7 x 10^{-3}	2.4 x 10^{-3}
Food	4.5 x 10^{-5} to 3.0 x 10^{-3} μg/g (depending on food type)	8.9 x 10^{-2}	3.3 x 10^{-2}
Total Estimated Daily Intake		1.1 x 10^{-1}	3.8 x 10^{-2}

Relative Rates of Absorption of Uranium into the Body

Relative rates of absorption, via ingestion, inhalation and dermal exposure, are required to derive an SQG_{HH}. These factors must be determined relative to the absorption rate in the toxicological study used to develop the TDI.

For ingestion and inhalation, a relative absorption rate of 1 (100%) was assumed for both the previous and revised CCME approaches. Although there may be evidence available that indicates that ingestion absorption of uranium in soil would be appreciably lower than uranium in drinking water (the exposure vehicle for the critical toxicological study upon which the TDI is based [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 skin (AF_S), this factor was assumed to be 0.05 (5%) for both CCME approaches. 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%). 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, the more conservative MDEP value of 0.05 (i.e., 5%) was used for both the CCME (1996) and revised draft (2005) CCME approaches.

Exposure Term

The exposure term (ET) is used to define the fraction of time that a person is assumed to spend at the site. In the CCME (1996) approach, it was assumed that persons would spend the following amounts of time at the site.

Agricultural/parkland/residential = 1.0 (i.e., 24 hr/day, 365 days/yr)
Commercial/industrial = 0.27 (i.e., 10 hr/day, 5 days/week, 48 weeks/yr)

Under the revised draft CCME approach, ET is divided into 2 terms: ET_1 (fraction of days per year at the site) and ET_2 (fraction of hours per day at the site). According to Health Canada guidance, the following assumptions are recommended:

ET_1: Agricultural/parkland/residential = 1.0 (i.e., 365 days/yr)
Commercial = 0.66 (i.e., 5 days/week, 48 weeks/yr)
Industrial = 0.66 (i.e., 5 days/week, 48 weeks/yr)

ET_2: Agricultural/parkland/residential = 1.0 (i.e., 24 hr/day)
Commercial = 0.41 (i.e., 10 hr/day)
Industrial = 0.41 (i.e., 10 hr/day)

IV. RESULTS AND DISCUSSION

Based on the equations and input parameters discussed above, generic SQG_{HH} were calculated for uranium using the CCME (1996) methods as well as the revised draft CCME approach. The estimated uranium SQG_{HH} for the various land uses are provided in Table 4 below.

Table 4: Calculated Generic SQG_{HH} Using the CCME (1996) and Revised Draft CCME Approaches

Land use	Possible SQG_{HH} (µg/g)	
	CCME (1996) Approach	Revised Draft CCME Approach
Agriculture	8	22
Residential/Parkland	8	22
Commercial	23	32
Industrial *	130 (due to off-site migration check)	280 (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.

It is apparent that application of the revised draft CCME approach results in an appreciable increase in the SQG_{HH} for uranium. The analysis indicates that the primary reason for the increased values is due to the adoption of Health Canada (2003) receptor characteristics, in particular the guidance for estimation of exposure from dermal contact with soil and, more specifically, the adoption of lower soil adherence factors. Under CCME (1996) approach, the soil adherence factors were an order of magnitude greater than those recommended by Health Canada (2003) and adopted within the 2005 revised protocol. The revised soil adherence factors are based on more recent scientific data provided by Kissel et al. (1996; 1998).

A second factor that had an influence on increasing the SQG_{HH} was the increase in body weight assumed for the toddler. Based on more recent Canadian data (see Richardson, 1997) Health Canada (2003) now recommends a mean body weight of 16.5 kg for toddlers when conducting contaminated site risk assessments.

It is important to note that not all of the revised draft CCME methods are less conservative than the previous CCME (1996) approach. In the case of soil ingestion (which is the most important pathway under the revised draft approach), these exposures incorporate an exposure term that is more conservative than previously used by CCME (1996) for commercial and industrial exposures. More specifically, Health Canada (2003) recommended that soil ingestion related exposures only be multiplied by the fraction of days per year at the site (i.e., 0.71 for commercial exposures) rather than the CCME (1996) approach that multiplied soil ingestion exposure by the typical fraction of hours per day at the site (i.e., 0.24 for commercial exposures).

The revised draft SQG_{HH} are considered to be adequately protective when applied on a generic basis. According to our estimates and using Health Canada (2003) guidance, a toddler exposed to a uranium soil concentration of 22 μg/g at a residence would have a uranium exposure of about 0.1 μg/kg bw/day which is about equivalent to the exposure that a young child would receive from the typical background food supply (see Table 3). In addition, such exposures from the residential SQG would yield an exposure that is still only 16% of the Health Canada TDI, 3% of the US EPA RfD and only 5% of ATSDR MRL. Furthermore, it is noted that Morris and Meinhold (1995; 1998) provide a probabilistic model that estimates exposures in the range of those from the PSQGHH (i.e., 0.1 mg/kg bw/day or less) would be associated with kidney concentrations less than 0.01 mg/g (i.e., more than 2 orders of magnitude lower less than the tissue concentration of 1 mg/g 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 PSQGHH would result in kidney uranium concentrations less than the 0.01 mg/g. Based on the above and other factors, the hypothetical SQG_{HH} derived herein should be conservative.

It is noted that garden produce and drinking water consumption have not been evaluated in this assessment. At sites where appreciable amounts of garden produce are consumed or where drinking water is sourced from nearby wells, a lower value may need to be considered. However, these are site-specific considerations to be addressed at specific contaminated sites, not in a generic guideline.

Finally, Health Canada is considering dermal bioavailability studies of uranium in soil (M. Richardson, pers. com.). 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.

V. CONCLUSIONS

Hypothetical generic SQG_{HH} were derived for uranium at contaminated sites using the revised draft CCME approach and compared to values that would have been obtained using the CCME (1996) methods. Although the SQG_{HH} will increase using the revised draft approach, they are considered to be protective of human health for the majority of Canadians. Nevertheless, as additional toxicological and other data become available, the SQG_{HH} should be re-evaluated to ensure adequate protection of human health.

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