CCME

Canadian Council of Ministers of the Environment | Le Conseil canadien des ministres de l'environnement

SCIENTIFIC CRITERIA DOCUMENT FOR THE DEVELOPMENT OF THE CANADIAN SOIL QUALITY GUIDELINES FOR THE PROTECTION OF HUMAN HEALTH

Lead

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LIST OF FREQUENTLY USED ACRONYMS AND ABBREVIATIONS

AF[1]	relative absorption factor
AFN	Assembly of First Nations
ATSDR	Agency for Toxic Substances and Disease Registry
BC MOE	British Columbia Ministry of Environment
BMD	benchmark dose
BLL	blood lead levels
bw	body weight
CalEPA	California Environmental Protection Agency
CalOEHHA	California Office of Environmental Health Hazard Assessment
CCME	Canadian Council of Ministers of the Environment
CCREM	Canadian Council of Resource and Environment Ministers
CDC	United States Centers for Disease Control and Prevention
CEPA	Canadian Environmental Protection Act
CMHC	Canadian Mortgage and Housing Corporation
CSoQG	Canadian Soil Quality Guidelines
d.w.	dry weight
EC	Environment Canada
ECCC	Environmental and Climate Change Canada
EDTA	ethylenediaminetetraacetic acid
EFSA	European Food Safety Authority
ERG	electroretinogram
HC	Health Canada
HWC	Health and Welfare Canada
ILZRO	International Lead Zinc Research Organization
INSPQ	Institut national de santé publique du Québec
IPCS	International Programme for Chemical Safety
IQ	intelligence quotient
LOD	limit of detection
NAPS	National Air Pollution Surveillance Program
NAS	National Academy of Science
NCRMP	National Chemical Residue Monitoring Program
NHANES	National Health and Nutrition Examination Survey
NIST	National Institute of Standards and Technology

[1] Note: The CCME protocol for the derivation of soil quality guidelines (CCME 2006) uses the acronym AF for relative absorption factor. In this document, RAF is used for relative absorption factor, except for in the equations where AF is used for consistency with the protocol.

NPRI	National Pollutant Release Inventory
NRCan	Natural Resources Canada
NTP	National Toxicology Program
OECD	Organisation for Economic Co-Operation and Development
OMEE	Ontario Ministry of the Environment and Energy
OMOE	Ontario Ministry of the Environment
Pb	lead
ppm	parts per million (e.g., µg/g, mg/kg, mg/L)
PSoQG	provisional soil quality guideline
$PSoQG_{DH}$	Provisional direct health-based soil quality guideline
$PSoQG_{HH}$	provisional soil quality guideline for human health
RAF	relative absorption factor
RSD	risk-specific dose
SBP	systolic blood pressure
SIR	soil ingestion rate
SoQG	soil quality guideline
$SoQG_{DH}$	soil quality guideline for direct human contact
$SoQG_{E}$	soil quality guideline for the protection of the environment
$SoQG_{F}$	final soil quality guideline
$SoQG_{FI}$	soil quality guideline for produce, meat and milk check
$SoQG_{FL}$	soil quality guideline for the protection of freshwater life
$SoQG_{HH}$	soil quality guideline for the protection of human health
$SoQG_{I}$	soil quality guideline for ingestion of soil and food
$SoQG_{IAQ}$	soil quality guideline for the protection of indoor air quality
$SoQG_{IR}$	soil quality guideline for the protection of irrigation water
$SoQG_{LW}$	soil quality guideline for the protection of livestock watering
$SoQG_{NEC}$	soil quality guideline nutrient and energy cycling check
$SoQG_{OM-E}$	soil quality guideline for the protection of environmental health off-site migration check
$SoQG_{OM-HH}$	soil quality guideline for human health for off-site migration check
$SoQG_{PW}$	soil quality guideline for the protection of potable water
$SoQG_{SC}$	soil quality guideline for soil contact
t	metric tonne (1,000 kg)
TDI	tolerable daily intake
TRV	toxicological reference value
UCLM	upper confidence level of the mean
UdM	Université de Montréal
U Ottawa	University of Ottawa

UNBC	University of Northern British Columbia
US EPA	United States Environmental Protection Agency
US HUD	United States Department of Housing and Urban Development
w.w.	wet weight
WHO	World Health Organization
WHO/JECFA	World Health Organization/Joint FAO/WHO Expert Committee on Food Additives

EXECUTIVE SUMMARY

Canadian environmental quality guidelines, developed by the Canadian Council of Ministers of the Environment (CCME), are numerical concentrations or narrative statements recommended to provide a healthy, functioning ecosystem capable of sustaining the existing and likely future uses of the site by ecological receptors and humans. Canadian Soil Quality Guidelines (CSoQG) can be used as the basis for consistent assessment and remediation of contaminated sites in Canada.

The guidelines were derived according to procedures described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (the Protocol, CCME 1996*a*, revised in 2006). According to the Protocol, both soil quality guidelines for the protection of the environment ($SoQG_E$) and human health ($SoQG_{HH}$) are developed, and the lowest value generated by the two approaches for each of the four land uses is recommended by CCME as the CSoQG (CCME 2006). This scientific criteria document provides the background information and rationale for the derivation of soil quality guidelines for lead (Pb) for the protection of human health only. This document contains a review of information on the chemical and physical properties of lead, a review of sources and emissions in Canada, the distribution and behaviour of lead in the environment, and the toxicological effects of lead on humans and laboratory animals. This information is used to derive soil quality guidelines for lead to protect two population age groups, toddlers and adults, in four types of land uses: agricultural, residential/parkland, commercial and industrial.

Based on the Protocol, three types of human health exposure pathways are evaluated: required pathways (direct contact via ingestion, inhalation and dermal contact); applicable pathways (indoor air; groundwater); and check mechanisms (off-site migration of substances; produce, meat and milk ingestion). The lowest value calculated for all applicable pathways is selected as the soil quality guideline for human health.

The provisional soil quality guidelines (PSoQGs) for human health ($PSoQGs_{HH}$) derived for lead for direct contact with soil via accidental ingestion, inhalation, and dermal contact are available for two risk levels. According to the Protocol, soil quality guidelines (SoQGs) are typically conservative and are intended to be protective of the majority of individuals in an exposed population. In the case of lead, the chosen risk levels for the derivation of the PSoQG are to protect for neurodevelopmental effects on a population basis, as it is not possible to identify such a low level of effects on an individual basis due to the influence of confounding variables and the related variability in intelligence quotient (IQ) test results. Since a threshold for effects cannot be determined for lead, two $PSoQGs_{HH}$ are developed based on two risk specific doses (RSD) (associated with decrements of 1- and 0.5- IQ points), to enable individual jurisdictions the opportunity to determine their science policy position and to allow for screening of soils at sites where exposure to other affected media may occur (Table 1). The level of protection associated with these values relates to soil concentrations resulting in no more than a 1-IQ point decrease on a population level. The data used to calculate the PSoQG for lead for the protection of human health are based on epidemiological studies, which may be influenced by a number of confounding factors, resulting in a variable level of protection within the population. For convenience, CCME (1999*a*) $SoQGs_E$ have also been provided in the tables of SoQGs for lead (Tables 1 and 3).

Table 1. Soil quality guidelines for lead (milligrams per kilogram [mg/kg])

	Land use			
	Agricultural	**Residential/ parkland**	**Commercial**	**Industrial**
$PSoQG_F$[c]				
0.5-IQ pt decrement[a]	61	61	82	600
1-IQ pt decrement[b]	70	113	154	600
$PSoQG_{HH}$				
0.5-IQ pt decrement[a]	61	61	82	743
1-IQ pt decrement[b]	113	113	154	1,477
Limiting pathway for $PSoQG_{HH}$	Direct contact	Direct contact	Direct contact	Off-site migration
$SoQG_E$	70	300	600	600
Limiting pathway for $SoQG_E$	Soil and food ingestion	Soil contact	Soil contact	Soil contact

Notes: $PSoQG_F$ = provisional final soil quality guideline; $PSoQG_{HH}$ = provisional soil quality guideline for human health; $SoQG_E$ = soil quality guideline for environmental health. Soil guidelines and the data used to calculate them are, by convention, always expressed on a dry weight basis to allow the data to be standardized. In case of doubt and if the scientific criteria document does not specify whether wet or dry weight is used, readers are advised to check the references provided.

[a] CCME recommends using the $PSoQG_{HH}$ for a 0.5-IQ point decrement when soil and additional site-related media contain elevated concentrations of Pb (e.g., groundwater, food grown on site, etc.) to account for additional sources of elevated exposure and/or according to jurisdictional policy.

[b] CCME recommends using the $PSoQG_{HH}$ for a 1-IQ point decrement when soil is the only site-related media with elevated concentrations of Pb. Where additional site-related contaminated media are expected to contribute to exposures (e.g., groundwater, food grown on site, etc.), CCME recommends using the 0.5-IQ point decrement to account for additional sources of elevated exposure. When using the $PSoQG_{HH}$ based on 1-IQ point decrement, the environmental site investigation report should include information on all media that may be affected above background and fate and transport information.

[c] Data are sufficient and adequate to calculate a $PSoQG_{HH}$ and an $SoQG_E$, the lower of which becomes the $PSoQG_F$ for each land use.

1. INTRODUCTION

Canadian Environmental Quality Guidelines are generic numerical concentrations or narrative statements intended to protect, sustain and enhance the quality of the Canadian environment, its many beneficial uses, and human health. These guidelines are developed using formal protocols to ensure consistent, scientifically defensible values. These values are endorsed across the country through CCME and are recommended for substances and other parameters (e.g., nutrients, pH) of concern in the environment.

The development of CSoQGs was initiated by CCME in 1991. In response to the urgent need to begin remediation of high priority "orphan" contaminated sites, CCME adopted an interim set of soil quality criteria from values used in various jurisdictions across Canada (CCME 1991). Since 1996, $CSoQGs_{HH}$ and $CSoQGs_{E}$ have been developed according to procedures described in the Protocol. According to the Protocol, 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 CSoQG.

CSoQGs for lead (Pb) were developed using the Protocol in 1997 (CCME 1997) and were revised and re-published in 1999 (CCME 1999*a*, Environment Canada [EC] 1999). The current revision (this document) to the CSoQGs for lead revises the $SoQG_{HH}$ only; however, these revised guidelines are provisional. The Pb $SoQG_{E}$ have not been revised since the 1999 publication.

Health Canada (HC) completed an assessment of the science on lead and consolidated the information in a State of the Science (SOS) Report (HC 2013*a*). The principal finding of this assessment is that critical health effects can occur at blood lead levels (BLLs) below 10 micrograms per deciliter (µg/dL). Consistent with conclusions from the larger scientific community, the SOS Report concludes that currently available observational studies do not demonstrate a population threshold for the most sensitive endpoint identified (neurodevelopmental toxicity), and therefore recommends additional measures to further reduce Canadian exposures to lead. The SOS Report conclusion forms the basis of this revision to the Canadian $SoQG_{HH}$ for lead.

This scientific criteria document contains a review of the chemical and physical properties of lead, a review of sources and emissions in Canada, the distribution and behaviour of lead in the environment, and the toxicological effects of lead on human populations. The $PSoQGs_{HH}$ for lead are based on a conservative receptor (a toddler) for agricultural, residential and commercial land uses, and an adult receptor (women of childbearing age, using the toddler toxicological reference value [TRV] to protect the developing fetus) for industrial land uses. Potential human exposure pathways for lead include direct contact with soil (ingestion, inhalation of fugitive dust from soil, and dermal contact). In addition, various check mechanisms are incorporated to consider indirect pathways of exposure (e.g., off-site migration of contaminants via wind and water erosion) and to provide protection for resources and receptors not otherwise considered in the derivation of SoQGs. However, the guideline for the protection of potable groundwater and the check to assess the transfer of contaminants from soil into produce, meat, and milk were not calculated. Effects on environmental receptors (e.g., microbes, plants, and wildlife) are not discussed in this scientific criteria document but can be found in EC (1999).

Under certain circumstances, site-specific conditions may require a deviation from default assumptions and associated $PSoQG_{HH}$. For example, the bioavailability of lead in soil can vary significantly as a function of the mineral phase, such as iron/lead sulphate, cerussite, native lead, organic lead and galena (Casteel *et al.* 2006). In some instances, collecting site-specific data may improve guideline accuracy on a regional basis or a contaminant-source basis. Furthermore, as a result of variability in geological conditions, it is possible that natural background concentrations may be higher than the $PSoQG_{HH}$. The reader should therefore consult appropriate federal, provincial or territorial regulatory authorities for additional guidance related to the development of site-specific SoQGs (e.g., CCME 1996*b*; 2006).

The $SoQGs_{HH}$ provide conservative (upper-bound) estimates of associated risk so that the guidelines are considered applicable to any area in Canada. Exposure parameters are typically a combination of central tendency (e.g., arithmetic mean or median) and upper-bound conservative values. For example, the Protocol uses mean body weights and applies upper-bound time activity patterns (hours/day and days/year of exposure). Exposure limits are typically conservative and intended to be protective of the majority of individuals in an exposed population.

The derivation of the lead PSoQG in this document follows the Protocol, with modifications to address several unique considerations with respect to this substance:

1. Several agencies have concluded that a threshold for effects cannot be determined for lead, and it is widely considered to be a non-threshold substance (California Environmental Protection Agency [CalEPA] 2009; European Food Safety Authority [EFSA] 2013; HC 2013*a*; US Environmental Protection Agency [US EPA] 2006*a*; Joint World Health Organization [WHO]/ Food and Agriculture Organization [FAO] Expert Committee on Food Additives [WHO/JECFA] 2011).
2. The RSDs used to develop the $PSoQGs_{HH}$ are based on neurodevelopmental effects in young children on a population basis (EFSA 2013; WHO/JECFA 2011).
3. Since a threshold for effects cannot be determined for lead, two $PSoQGs_{HH}$ are developed based on EFSA (2013) to provide two toddler RSD levels (targeting decrements of 1- and 0.5-IQ points), to enable individual jurisdictions the opportunity to determine their science policy position.
4. Separate RSDs are provided for young children and adults; as the adult RSD may not specifically protect the developing fetus at sites with women of childbearing age, all guidelines are developed based on the RSDs to protect young children.
5. The off-site migration check for industrial land uses was considered protective of the potential for windblown dust from industrial sites to affect the soil concentrations of nearby agricultural and residential lands. However, the off-site migration check does not address situations where soils from industrial sites may remain on shoes/clothes and potentially result in elevated lead levels of indoor settled dust in an off-site home, where children may come into contact with it. In cases where this situation may be a concern, a site-specific assessment is recommended.
6. Since the guideline for the protection of potable groundwater and the check to assess the transfer of contaminants from soil into produce, meat, and milk were not calculated, the $PSoQG_{HH}$ based on the protection of a 0.5-IQ point decrement is recommended for screening purposes, where exposure to other affected media at the site may occur.

Much of the background information in this report has been extracted from previous reports prepared by, or on behalf of, the Contaminated Sites Division of HC (Damman *et al.* 2005; Wilson *et al*. 2005; Equilibrium Environmental Inc. 2008*a*; *b*; Healey *et al*. 2010; SENES Consultants Ltd. 2010).

2. BACKGROUND INFORMATION

2.1 Physical and Chemical Properties

Lead is an odourless, bluish-gray, lustrous metal that is malleable, ductile and resistant to chemical corrosion. Lead (CAS No. 7439-92-1) is a post-transition metal in Group IVA (14) of the Periodic Table, with an atomic number of 82 and an atomic weight of 207.2 (O'Neil 2001). It has a melting point of 327.4 °C and a boiling point of 1,740 °C at atmospheric pressure (Agency for Toxic Substances and Disease Registry [ATSDR] 2007*a*). Table 1 presents some of lead's physical and chemical properties.

Elemental lead exists in three oxidation states (Pb^0, Pb^{2+} and Pb^{4+}), and the most common valence states found in the environment are Pb^{2+} and Pb^{4+} (Canadian Council of Resource and Environmental Ministers [CCREM] 1987; Reimann and de Caritat 1998). There are also four stable (non-radioactive) isotopes: ^{204}Pb, ^{206}Pb, ^{207}Pb and ^{208}Pb. The water solubility of lead salts ranges broadly and depends on the associated anions. For example, lead nitrate and acetate are water soluble, whereas lead carbonate, phosphate, sulphate and sulphide are relatively insoluble (Ontario Ministry of the Environment and Energy [OMEE] 1994). Lead can form chelated compounds with calcium disodium ethylenediaminetetraacetic acid (EDTA) (used to treat lead poisoning), as well as with various nucleotide- and peptide-containing compounds (OMEE 1994). Lead readily alloys with other metals such as tin, antimony, copper and zinc.

2.2 Geochemical Occurrence

Of the heavy metals with an atomic number > 60, lead is the most abundant in the earth's crust (Adriano 2001; WHO 2010). Lead occurs naturally in bedrock, soils, tills, sediments, surface waters, groundwater and seawater (Reimann and de Caritat 1998). As a result, it also occurs naturally at low levels in foods due to uptake from soil into plants, particulate deposition onto plants, uptake from water and sediments into fish, and uptake into animals that consume plants and other animals (Adriano 2001).

Elemental (native) lead is rare in nature, where it predominately exists in the divalent (plumbous) state in combination with organic and inorganic compounds such as galena (PbS; lead sulphide), anglesite ($PbSO_4$; lead sulphate), cerussite ($PbCO_3$; lead carbonate), pyromorphite ($Pb_5(PO_4)_3Cl$; leadchlorophosphate) and mimetesite ($Pb_5(AsO_4)_3Cl$; lead arsenate chloride) (ATSDR 2007*a*; Reimann and de Caritat 1998). Lead coexists in ore deposits with other metals, particularly zinc, copper and cadmium (Adriano 2001; Reimann and de Caritat 1998).

Table 2. Physical and chemical properties of some lead compounds*

Property	Compound										
	Elemental lead	**Galena ore**	**Lead oxide**	**Lead dioxide**	**Lead (II) acetate**	**Lead arsenate**	**Lead phosphate**	**Lead sulphate**	**Lead chloride**	**Lead carbonate**	**Tetraethyl lead**
Chemical formula	Pb	PbS	PbO	PbO_2	$Pb(CH_3CO_2)_2$	$PbHAsO_4$	$(Pb_3(PO_4)2)$	$PbSO_4$	$PbCl_2$	$PbCO_3$	C_8H_2OPb
CAS registry number	7439-92-1	1314-87-0	1317-36-8	1309-60-0	301-04-2	7784-40-9	7446-27-7	7446-14-2	7758-95-4	598-63-0	78-00-2
Molecular weight g/mol	207.2	239.27	223.0	239.20	325.29	347.1	811.54	303.26	278.11	267.2	323.44
Physical state (@ 25 ºC)	Solid metal	Metallic black cubic crystals	red to reddish-yellow tetragonal crystals & yellow orthorhombic crystals	Dark-brown powder	White crystals	White, heavy powder	White powder	White heavy crystal powder	White crystalline powder	Colourless rhombic crystals	Colourless oily liquid
Melting point (ºC)	327.4	1,114	888	290 (decomp)	280	280 (decomp)	1,014	1,170	501	315 (decomp)	-133.7
Boiling point (ºC)	1,740	1,281 (sublimes)	N/A	N/A	decomp	N/A	N/A	N/A	950	N/A	200
Density (g/cm^3 @ or near room temperature)	11.34	7.57–7.59	9.53	9.38	3.25	5.943	6.9	6.2	5.85	6.582	1.653
Water solubility	Insoluble	0.000086 g/100 mL @ 13 °C	0.0504 g/L (alpha form); 0.1065 g/L (beta form)	Insoluble	44.3 g/100 mL @ 20 °C	Insoluble	0.000014 g/100 mL @ 20 °C	0.0404 g/100 mL @ 25 °C	Soluble in 93 parts cold water, 30 parts boiling water	0.00011 g/100 mL @ 20 °C	0.29 mg/L @ 25 °C

decomp = decomposition; N/A = not available; * Source: Hazardous Substance Data Bank (HSDB) 2010.

2.3 Analytical Methods

Several North American regulatory agencies recommend methods to analyse lead in various matrices. CCME (2016) published guidance on analytical methods for the characterization of contaminated sites that indicate the preferred methods for those matrices commonly sampled at contaminated sites.

Prior to analysis, the solubilization of total lead in solid matrices can be achieved with a strong acid extraction. Strong acid digestion (e.g., EPA Method 3050B: Acid Digestion of Sediments, Sludges and Soils; US EPA 1996*a*) can help determine the amount of available lead bound to soil or dust. The nitric-hydrochloric acid digest (such as EPA Method 3050B) is typically used for environmental analysis of soils. A stronger digest, such as hydrofluoric and/or perchloric acid (e.g., EPA Method 3052; US EPA 1996*b*), would result in the release of all lead from soils, including lead bound in a crystalline/silica matrix, which is not typically available when soils are ingested. This type of digest (referred to as a total, rather than available metals, digest) is more commonly used in conjunction with geological exploration than environmental studies. Several other methods have been developed to better approximate the amount of lead that may be accessible for oral absorption; they are described in Section 2.3.1.

2.3.1 In Vitro Bioaccessibility Tests

Various *in vitro* bioaccessibility methods have been developed to estimate the oral bioavailability of metals, as this can vary from soil to soil, depending on a number of factors, including soil physical-chemical properties, metal speciation, weathering, etc. HC (2017*b*) provides additional detail on this topic.

The relative bioavailability often represents the bioavailability of lead in soil relative to the bioavailability of a soluble form (e.g., lead acetate). If bioavailability is going to be assessed at a site, consult guidance from other jurisdictions (e.g., HC 2017*b*; US EPA 2012) for guidance on how to conduct assessments.

2.4 Production and Uses in Canada

Both soluble and insoluble lead compounds have a variety of industrial uses and have been intentionally added to a broad range of products including, but not limited to, plumbing pipes and fixtures, batteries, paint and plastics, cable sheathing, circuit boards, lining for chemical baths and storage vessels, chemical transmission pipes, decorative and optical glass, electrical components and curtain weights (National Toxicology Program [NTP] 2004). Lead is used extensively in rolled and extruded lead products in the construction industry (International Agency for Research on Cancer [IARC] 2006).

Soluble lead compounds are used in diverse industries. Lead acetate is used as a water repellent for mildew protection and as a mordant for dyes. Lead acetate trihydrate is used to produce varnishes and chrome pigments, and as an analytical reagent, while lead chloride is used to manufacture asbestos clutch or brake linings, and as a catalyst and flame retardant. Lead nitrate is

used to manufacture nylon, matches and explosives, and as a coating on paper for photothermography.

Insoluble lead compounds are just as widely used. Lead azide and lead styphnate are used to manufacture munitions. Lead carbonate, lead fluoride, lead fluoroborate and lead naphthenate are employed as catalysts. They are also used in the electronic and optical industries (lead fluoride), in coatings for thermographic copying (lead carbonate), in epoxy resins (lead fluoroborate), and in varnish (lead naphthenate). Both lead phosphate and lead stearate are used as stabilizers in the plastics industry. Lead iodide is used in thermoelectric materials and lead sulphate in the production of galvanic batteries, and both were previously used in photography. Lead oxide and lead sulphide are used in ceramic glazes, as well as to vulcanize rubber and plastics (lead oxide) and to sense humidity in rockets (lead sulphide). Until the 1960s, significant quantities of lead (10–50%) were added to paints, rubber and plastics either as a pigment (lead (II) chromate) or to speed drying, resist corrosion and increase durability (lead (II) carbonate), while lead tetraoxide was used in plasters, ointments, glazes and varnishes (ATSDR 2007*b*; Canadian Mortgage and Housing Corporation [CMHC] 2004). Lead thiocyanate is used to manufacture matches and cartridges, while lead arsenate was historically used as an insecticide and herbicide but has no current application (NTP 2004).

Organic lead compounds, including tetraethyl lead and tetramethyl lead, were once widely used in fuels prior to their total prohibition in on-road vehicles in North America in the 1990s. Presently, in Canada, these compounds have limited approved uses in piston engine aircraft (aviation gasoline [avgas]) and racing fuel for competition vehicles[2], as an exemption in the *Gasoline Regulations* (Canadian Environmental Protection Act [CEPA] 1999).

Canada is a significant global producer and supplier of refined lead, ranking eighth in the world in 2018 in terms of mine production (13 897 tonnes) and eighth in terms of refined lead production (255 245 tonnes) (Natural Resources Canada [NRCan] 2018). Most lead in Canada is produced as a co-product of zinc mining, while lead recycling, mainly from depleted car batteries, represented the primary source of Canada's total refined production (45%) in 2018. Nearly 90% of refined lead produced in Canada is exported to the United States. In 2018, primary refined lead metal was produced using domestic and foreign concentrates at two smelters located in New Brunswick and British Columbia, while secondary lead metal was produced from recycled lead (primarily car batteries) at three sites in Québec, Ontario and British Columbia, in addition to the primary lead smelters (NRCan 2018). The Belledune smelter in New Brunswick closed in 2019.

Given its ubiquitous distribution, historic use and toxicity, numerous worldwide regulations cover the use of lead in order to limit the risks to human health and the environment. HC (2013*b*) provides a summary of Canadian regulations regarding lead.

[2] A competition vehicle is defined in the *Gasoline Regulations* as "a vehicle or boat that is used exclusively for competition and does not include a vehicle that is used on a highway or a vehicle or boat that is used for recreational purposes."

2.5 Sources and Concentrations in the Canadian Environment

High concentrations of metals can occur naturally in soils, sediments and water due to natural processes, which can blur the distinction between anthropogenic pollution and naturally occurring lead (EC 1996). 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 above what is considered typical Canadian background. Lake or stream sediments act as sinks, accumulating elements from surrounding watersheds (i.e., lead within bedrock, glacial sediments and soils).

Numerous anthropogenic sources produce lead. According to Environment and Climate Change Canada (ECCC)'s National Pollutant Release Inventory (NPRI), approximately 223,000 kg of lead and lead compounds were released into the Canadian environment in 2019 (93,000 kg to air, 9,800 kg to water and 120,000 kg to land) (ECCC 2020). Approximately 60% of air emissions in 2017 were released by the mining and metals production industries, and Canadian military bases accounted for approximately 96% of the releases to land (ECCC 2018). However, NPRI reporting is required for those organizations that meet reporting criteria (EC 2012), and it does not represent all industrial releases or sources (e.g., lead in products such as shot and sinkers).

Prior to the interdiction of lead in gasoline in the 1990s, anthropogenic lead emissions were estimated to exceed natural emissions by one to two orders of magnitude (Flegal *et al.* 1990; Jaworski *et al*. 1987). Excluding mobile sources, global anthropogenic atmospheric emissions were estimated to range from 72.2 to 94 kilotonnes/year (Nriagu and Pacyna 1988). Water discharges were estimated to range from 10 to 67 kilotonnes/year, and soil discharges were estimated to range from 606 to 1,630 kilotonnes/year (EC 1982). Most environmental impacts of lead tend to be relatively localized (Ewers and Schipköter 1991).

In countries that have ceased to use leaded gasoline, releases from the non-ferrous smelting and refining industry and the recycling or disposal of products containing lead (ATSDR 2007*a*; ECCC 2020; International Programme for Chemical Safety [IPCS] 1995) represent the primary anthropogenic emissions of lead (United Nations Environment Programme 2010). In Canada, avgas is the third-largest source of lead emissions (ECCC 2020). Lead emissions from competition vehicles (such as Formula 1 cars) represent 0.3% of Canadian emissions (Government of Canada 2010). Electrical utilities also release lead into the environment in flue gas by burning lead-contaminated fuels, such as coal (ATSDR 2007*a*).

Lead is listed on Schedule 1, List of Toxic Substances, under CEPA 1999.

2.5.1 Ambient Air

Atmospheric lead is mainly associated with aerosol particles < 1 µm in diameter (Sannolo *et al*. 1995*b*), generated predominantly by anthropogenic high-temperature processes such as smelting and incineration (Bennett and Knapp 1989; Hill 1992; Jaworski *et al*. 1987). Air quality data for lead in particulate matter with diameters < 10 µm (PM_{10}) and < 2.5 µm ($PM_{2.5}$) are available from 26 sites across Canada via ECCC's National Air Pollution Surveillance program (NAPS). Between 2000 and 2009, the 5^{th} to 95^{th} percentile concentrations of lead ranged from 0.0004 to 0.014 µg/m^3

in $PM_{2.5}$ (EC 2010). Higher concentrations have been reported in total suspended particulates (TSP), $PM_{2.5}$ and PM_{10} in the vicinity of industrial sources (Brecher *et al*. 1989; Dobrin and Potvin 1992; OMEE 1992). For example, the maximum recorded NAPS lead concentration (0.5981 $\mu g/m^3$) was from a sample collected near Flin Flon, Manitoba, in 2007[3] (EC 2010).

Particulate NAPS data demonstrate that concentrations declined significantly following the introduction of unleaded gasoline in 1975 and the prohibition of leaded gasoline in on-road vehicles in the 1990s. In Canada, ambient air lead concentrations declined by > 99% between 1984 (0.16 $\mu g/m^3$) and 2008 (< 0.0015 $\mu g/m^3$) (Government of Canada 2020*a*). The prohibition of leaded fuels, combined with the imposition of greater controls on lead mining and smelting emissions, resulted in average ambient lead concentrations consistently < 0.02 $\mu g/m^3$ (HC 2013*a*). In matched indoor, outdoor and personal $PM_{2.5}$ samples from Windsor, Ontario, median lead content ranged from 0.001 to 0.010 $\mu g/m^3$ (limit of detection [LOD] = 0.002 µg lead/filter, 0.15 mg particles/filter) for samples collected between 2004 and 2006 (Rasmussen *et al*. 2007; 2009).

A summary of North American ambient air data is provided in Appendix 1.

2.5.2 Indoor Air

Indoor air lead data are summarized in Appendix 1, and Canadian data are minimal. Rasmussen *et al.* (2006) reported indoor air lead concentrations in homes of non-smokers in Ottawa, Ontario, in 2002. Concentrations in $PM_{2.5}$ (LOD 0.0002 $\mu g/m^3$) ranged from 0.0004 to 0.0027 $\mu g/m^3$ in rural residences (n = 10; median = 0.0023 $\mu g/m^3$) and from 0.0010 to 0.0051 $\mu g/m^3$ in urban residences (n = 10; median = 0.0015 $\mu g/m^3$).

The American National Human Exposure Assessment Survey (Clayton *et al.* 1999) reported a median indoor air lead concentration (from particles ≤ 50 µm in diameter) of 0.0066 $\mu g/m^3$ (n = 213).

2.5.3 Indoor Dust

Concentrations of lead in indoor settled dust have been correlated with outdoor soil near smelters, degraded indoor leaded paint, and potentially, tobacco smoke. Lead in soil was the primary underlying source of variation in indoor dust (Zahran *et al*. 2011), while Spalinger *et al.* (2007), Adgate *et al.* (1998*b*) and Meyer *et al.* (1999*a*; *b*) all showed associations between soil lead and indoor dust concentrations. Several analyses link lead in soil around the home or play area to increased exposure in children (Lanphear *et al*. 1998). Von Lindern *et al.* (2003) determined that BLLs of children living near a smelter were influenced mainly by house dust. House dust accounted for 42% of the soil/dust contribution to blood lead, with community soils contributing 27%, neighbourhood soils 19%, and residential yard soil an estimated 12% (von Lindern *et al.* 2003).

[3] This facility closed in 2010.

Several sources (Bushnik *et al*. 2010; Jacobs *et al.* 2002; United States Department of Housing and Urban Development [US HUD] 2001) confirmed that BLLs are significantly higher in individuals living in older homes (greater than 50 years) than in newer homes, due to the presence of lead-containing paints. Weathering or disintegration of such paints or dust generated during renovations can significantly increase dust lead levels (Farfel and Chisolm 1990; US HUD 2001). Various hobbies can also contribute to elevated household lead levels (Sanborn *et al.* 2002). The removal or remediation of lead sources can reduce indoor dust lead levels, which directly reduces BLLs in children (British Columbia Ministry of Environment [BC MOE] 2009; Hilts 2003; Lanphear *et al*. 2003; Rhoads *et al.* 1999).

Mixed results have been reported regarding the impact of smoking on dust lead levels. A German study suggested that smoking increases dust lead levels, while a Scandinavian study found no significant association between smoking and dust lead levels (Hogervorst *et al.* 2007; Willers *et al.* 1993).

Indoor dust lead concentrations were measured in several Canadian studies (Hilts 2003; Intrinsik Inc. 2010; McDonald *et al*. 2010; 2011; Rasmussen *et al*. 2001; 2011; 2013; Roy *et al*. 1993). Generally, higher lead concentrations were found in regions close to point sources (Hilts 2003; Intrinsik Inc. 2010; Roy *et al*. 1993) and there was also a moderate correlation between indoor dust lead concentrations (and bioaccessibility) and the age of homes, as older homes with lead paint often had higher concentrations than newer homes (although 10% of the homes with elevated dust lead concentrations were built after 1980) (McDonald *et al*. 2010; 2011; Rasmussen *et al*. 2013). Speciation studies to determine the bioaccessibility of lead in dust indicate that, besides both interior and exterior paint sources, lead may accumulate in indoor dust from a multitude of other possible sources, including dirt transferred indoors and consumer products (MacLean *et al.* 2011). Additionally, certain hobbies, such as making pottery, stained glass or fishing lures and sinkers, as well as refinishing furniture, can contribute to lead concentrations in the home, and activities outside the home may result in lead being tracked inside (McDonald *et al*. 2010).

A summary of these data is provided in Appendix 1.

2.5.4 Soil

Background Canadian soil lead concentrations available from the Geological Survey of Canada (GSC) generally reflect glacial till (Grunsky 2010; Rencz *et al*. 2006). The GSC database is the most populated Canadian soils database available, although it does not cover all regions or soil types. Reported lead concentrations in glacial till range from < 2.0 to 152 mg/kg (arithmetic mean = 10 mg/kg; 90th percentile = 16 mg/kg; n = 7,398; < 63 µm size fraction) (Grunsky 2010). Dodd *et al*. (2017) reported lead concentrations in soil collected from reference locations across Canada as part of the North American Soil Geochemical Landscape Project (also known as the Tri-National Survey). A mean lead concentration of 26 mg/kg (95% upper confidence level of the mean [UCLM] = 27 mg/kg; n = 532) was reported for soils in the top 5 cm (i.e., the public health layer: the soil to which humans are most often exposed). In contrast, lead concentrations were lower in C-horizon soils, with a mean of 12 mg/kg (95% UCLM = 13 mg/kg; n = 532).

Due to the historical dispersive uses of lead, most surface soils around the globe are enriched by anthropogenic lead, which can remain there indefinitely, as it is non-volatile, has low solubility and readily complexes with organic matter. While Canadian background data available from the GSC generally reflect glacial till, atmospheric deposition associated with anthropogenic activity has increased concentrations in the upper portion of soils in a manner reflective of human activity (Talbot *et al*. 2017). The main anthropogenic source of lead was leaded fuels (Talbot *et al*. 2017). Soil lead levels tend to be higher in cities, near roadways, in areas next to homes and buildings with lead paint (ATSDR 2007*a*; CMHC 2004; Krueger and Duguay 1989; Mielke *et al*. 1989; Organisation for Economic Co-operation and Development [OECD] 1993; Schmitt *et al*. 1988) and around industrial sources that use or emit lead, such as metal mining/smelting operations (Alloway 1995; Hilts 2003; Kabata-Pendias 2001; Kelly *et al*. 1991; von Lindern *et al.* 2003).

Rasmussen *et al*. (2001) reported lead concentrations ranging from 16 to 550 mg/kg (n = 48; median = 34 mg/kg) in urban Ottawa, Ontario, garden soils. High concentrations in the vicinity of industrial sites were measured near the Trail, British Columbia, smelter (geometric mean = 765 mg/kg) (Hilts 2003) and a lead-zinc smelter in Flin Flon, Manitoba (lead levels in the city soil = 5–1,400 mg/kg) (Manitoba Conservation 2007). Of 106 sites tested in Flin Flon and neighbouring Creighton, Saskatchewan, 41% exceeded 140 mg/kg (Manitoba Conservation 2007). Studies have demonstrated that exposure to lead, particularly in children, is often directly related to the concentration of lead in nearby soil (Lanphear *et al*. 1998; 2003; Spalinger *et al*. 2007; von Lindern *et al*. 2003).

For the purpose of deriving CSoQGs, CCME used the background concentration of lead in till obtained by the GSC to represent Canadian background concentrations (arithmetic mean concentration of 10 mg/kg). However, lead concentrations are typically elevated in urban surface soils due to historical activities, including the use of leaded gasoline, and it is likely that exposure to lead in surface soils would be higher than in till (e.g., mean = 26 mg/kg in surface soils). Additionally, background concentrations may be higher than the estimated Canadian background due to local geochemistry. Site-specific assessments may consider the local background concentrations of lead in soils to estimate the background soil concentration (BSC).

A summary of these data is provided in Appendix 1.

2.5.5 Surface Water and Sediments

Lead can enter water bodies and groundwater through the natural weathering of rocks and soil, indirectly from atmospheric fallout and deposition, or directly from industrial sources and underground infrastructure (ATSDR 2007*a*; IARC 2006). Surface water and sediment can also be affected by sewage releases, harbour activities, runoff from nearby lead storage and production sites such as smelting and refining plants (IARC 2006), and lead leaching from spent ammunition and discarded fishing weights (IARC 2006).

Lakes and rivers have been reported to have lead concentrations (total, including both dissolved and adsorbed forms) in the range of 0.1 to 10 µg/L, with much higher lead concentrations reported in waters affected by lead emissions and discharges (Ewers and Schipköter 1991; Mayer and

Manning 1990). Sediment from Tadanac Lake near Parry Sound, Ontario, an undisturbed Precambrian Shield lake with no direct anthropogenic sources of lead, contained reported average lead concentrations of 25 to 225 µg/g (dry weight [d.w.]) with a mean of 98 µg/g (d.w.); however, sediment samples from an area with suspected contamination in Sudbury, Ontario, had lead levels up to 2,228 µg/g (Crocket and Kabir 1981; Crowder *et al.* 1989; Hamdy and Post 1985; Johnson and Nicholls 1988; Lum and Gammon 1985; Mudroch 1991; OMEE 1994; Samant *et al*. 1990; Wren *et al*. 1983).

Overall, lead levels in both surface waters and sediments have decreased since the 1970s and 1980s, presumably due to decreased atmospheric deposition of lead from the combustion of leaded gasoline (Nelson and Campbell 1991; Siver and Wozniak 2001).

A summary of surface water and sediment data is provided in Appendix 1.

2.5.6 Drinking Water

Lead concentrations in source water (e.g., rivers, lakes and groundwater) are typically very low (HC 2019*a*). However, lead can be introduced into drinking water after it leaves treatment plants from lead service lines, connection pipes, internal domestic plumbing, storage tanks and plumbing fixtures (ATSDR 2007*a*; *b*; HC 2019*a*; Sannolo *et al.* 1995*a*).

Lead service lines have been shown to be consistently high sources of lead for many years after installation under various conditions (Britton and Richards 1981; Cartier *et al*. 2011; 2012; Sandvig *et al*. 2008; Schock *et al*. 1996; Xie and Giammar 2011). The amount of lead leaching from the plumbing depends on several factors, including the age of the plumbing (leaching decreases over time) (Boffardi 1988; 1990; Boyd *et al*. 2008; 2012; Zhang and Edwards 2011); water chemistry (e.g., water temperature, pH, buffering capacity/alkalinity) (Clark *et al.* 2014; Deshommes *et al*. 2012; Lee *et al*. 1989; Lytle and Schock 1996; 2000; Maas *et al*. 1991); the length of time the water sits in the pipes (HC 2009; Sandvig *et al*. 2008); and the season (higher lead concentrations are correlated with warmer summer temperatures) (Britton and Richards 1981; Colling *et al*. 1987; 1992; Douglas *et al*. 2004; Karalekas *et al*. 1983). Water systems with high oxidation-reduction potential (ORP) (e.g., high residual chlorine) may form stable lead dioxide scales. The introduction of chloriamine lowers ORP conditions and can cause the breakdown of the lead dioxide scales to soluble lead through reductive dissolution (Edwards *et al*. 2009). Lead concentrations can be significantly higher when lead is in the particulate form. Particulate lead can originate from lead solder particles originating from the distributions system. (HC 2019*a*). See HC (2019*a*) for more information on sources of lead in drinking water.

The concentration of lead can vary significantly both across a distribution system and at an individual site (American Water Works Association [AWWA] Research Foundation 1990; Bailey and Russell 1981; Karalekas *et al*. 1978; Schock 1990; Schock and Lemieux 2010), making it challenging to assess lead exposure from drinking water. Exposure can be properly assessed only by monitoring lead levels at the tap, because lead in tap water is principally a result of dissolution (corrosion) from components of plumbing systems. No Canada-wide database exists for lead

concentrations in Canadian drinking water; however, many municipalities and provinces maintain databases of the results of water quality analyses that include lead.

Concentrations of lead in drinking water distribution systems from the National Survey of Disinfection By-Products and Selected Drinking Water Contaminants in Canadian Drinking Water (2009–2010) (HC 2014), and available provincial/territorial data, are presented in Appendix 1.

2.5.7 Biota

Concentrations of lead in biota used as human food are summarized below, and in Appendix 1.

2.5.7.1 Biota Used as Human Food

Lead occurs in plants due to uptake from soil or from atmospheric deposition, and it can subsequently move up the food chain. Lead can also be taken up from water and sediment by aquatic species (Adriano 2001; IPCS 1995; US EPA 1986). Certain sub-populations may receive additional lead exposure through the consumption of wild game and fish. To that extent, the European Union banned lead from ammunition used in wetlands in 2019 and proposes the restriction of lead ammunition for hunting and recreation shooting as well as lead used in fishing applications (European Chemicals Agency [ECHA] 2021).

Lead levels in a variety of fish collected from Lakes Ontario, Erie and Superior ranged from < 1.8 to 96.7 ng/g (Forsyth *et al*. 1990). Hellou *et al*. (1992) reported concentrations of < 40 ng/g d.w. (< 7.2 ng/g wet weight [w.w.]) in muscle and < 100 ng/g d.w. (< 53 ng/g w.w.) in the liver of northwest Atlantic cod *(Gadus morhua)*. Lobel *et al*. (1991) reported lead levels from 120 ng/g (d.w.; foot) to 23,000 ng/g (d.w.; kidney) of mussels from Bellevue, Newfoundland and Labrador. Lead concentrations in BC spot prawns (*Pandalus platyceros;* abdominal tissue) ranged from 24 to 262 ng/g (d.w.; mean = 90 ng/g) (Whyte and Boutillier 1991).

High concentrations of lead have been measured in game animals such as deer, caribou, moose, rabbit, squirrel and game birds hunted using lead bullets (Fachehoun *et al*. 2015; Falandysz *et. al*. 2005; Lewis *et al*. 2001; Medvedev 1999; Rodrigue *et al*. 2005; Tsuji and Nieboer 1997; University of Ottawa [U Ottawa], Université de Montréal [UdM], Assembly of First Nations [AFN] 2014; 2016; 2017; University of Northern British Columbia [UNBC], UdM, AFN 2011; 2012). In Canada, lead has been found in many country foods such as fish and seafood, berries, fruit, mushrooms and vegetables (U Ottawa, UdM, AFN 2014; 2016; 2017; UNBC, UdM, AFN 2011; 2012). High concentrations of lead (range = 12.24 to 3,700 µg/g) have been observed in several game species of mammals and birds, most notably in the meat of large and small game such as deer (Atlantic provinces, Manitoba [MB], Alberta [AB] and British Columbia [BC]), moose (MB), bison (AB and BC), squirrel (Atlantic provinces), rabbit (MB and AB), and game birds and waterfowl (MB) (U Ottawa, UdM, AFN 2014; 2016; 2017; UNBC, UdM, AFN 2011; 2012; data presented in Appendix 1). Deer and caribou hunted in Northern Ontario had a range of tissue lead concentrations of 0.1 to 5,726 µg/g (Tsuji and Nieboer 1997), with the highest levels associated with the bullet wound region. Lead levels were not consistent in additional samples

obtained from the same animal, indicating that bullet fragmentation and the location of the wound can affect lead concentrations.

While Canada has prohibited the use of lead shot for hunting migratory game birds and waterfowl since 1999 under the *Migratory Birds Convention Act*, consumers of animals killed with lead ammunition can be exposed to elevated lead levels from bullet fragments remaining in game meat, even after commercial processing (Dobrowolska and Melosik 2008; Hunt *et al.* 2009; Iqbal *et al.* 2009; Pain *et al.* 2010).

Lead concentrations in traditional food samples consumed by BC First Nations living on-reserve were assessed as part of the First Nations, Food, Nutrition and Environment Study. Lead concentrations in all food items were at the background level except for beaver heart, Canada goose, deer and grouse meat; the highest concentration of lead was reported in grouse meat at 60 µg/kg, likely from lead shot (UNBC, UdM, AFN 2011).

2.5.7.2 Commercial Foods

Lead can be found in various foods grown in soil with naturally occurring lead, as well as soil contaminated with lead due to proximity to historical emission sources (such as lead smelting, mining and refining facilities) and areas with high automobile traffic (HC 2013*a*). Crops can also become contaminated by precipitation and atmospheric particulate deposition, in which particles cling to the plant's surfaces and are actively taken up and sequestered into internal plant tissues (German Federal Ministry for Economic Cooperation and Development 2010). Lead contamination can further be introduced into food and beverages by processing practices, during transport to market, by water used for cooking, or from utensils and storage vessels containing lead (HC 2013*a*). In general, lead concentrations in commercial foods have decreased over time, particularly since the phase-out of lead by the food processing industry and the switch from leaded to unleaded gas. HC provides a maximum level for lead in fruit juice, fruit nectar and water in sealed containers (Government of Canada 2017).

Lead levels from food samples collected from 2003 to 2018 from a wide variety of commercially available foods under HC's Total Diet Study (TDS) range from < 0.1 µg/kg in natural spring water to 639 µg/kg in herbs and spices. However, when herbs and spices and salt are removed from the data set, the range of concentrations decreases to < 0.1 µg/kg (in some fresh fruit and vegetable, oil products, sugar, soft drink, and water samples) to 83 µg/kg (in chewing gum) for the 2016-2018 TDS data set (Government of Canada 2020*b*). Overall, lead levels are consistently highest in herbs and spices and salt (HC 2013*a*; 2019*b*; Government of Canada 2020*b*). The food groups contributing most to the dietary intake of lead since 2004 in Canada are beverages (e.g., beer, wine, coffee, tea, soft drinks), cereal-based foods, and vegetables (HC 2013*a*). Cereal products are the largest lead contributor to European dietary exposure (EFSA 2013). Average levels of lead in Canadian commercial foods are presented in Appendix 1.

Lead levels measured in 836 processed food samples through the Canadian Food Inspection Agency's (CFIA) National Chemical Residue Monitoring Program (NCRMP) and Children's

Food Project (HC 2013*a*) were above detection (5 µg/kg) in 44% of grain-based products in 2007–2008.

2.5.7.3 Infant Formula and Human Breast Milk

Mobilization of lead from the mothers' bones during pregnancy and the ingestion of food contaminated with lead are the dominant sources of lead in breast milk (Gulson *et al.* 1998). A 2003 Canadian survey of Cree women found an average of 2.08 µg/L (range: 0.41–8.33 µg/L) (Hanning *et al*. 2003). A 1981 survey of breast milk from 210 mothers from across Canada measured lead concentrations of < 0.025 to 15.8 µg/L (arithmetic and geometric means = 1.04 and 0.566 µg/L, respectively) (Dabeka *et al*. 1986). Data on levels of lead in breast milk are summarized in Appendix 1.

Formula-fed infants can potentially be exposed to greater levels of lead than those consuming only breast milk due to the introduction of lead in tap water used to reconstitute formula, or from formulas that contain calcium derived from natural sources (ATSDR 2007*a*). Despite the potential introduction of lead from the aforementioned sources, in general, lead levels in infant formulas are similar to those in breast milk (Rabinowitz *et al*. 1985). HC measured mean lead levels from 0.09 ng/g (in 5% glucose formula) to 1.58 ng/g (in hypoallergenic milk-based powder) from a variety of infant formulas (HC 2010*b*).

2.6 Consumer Products

According to the International Agency for Research on Cancer (IARC 2006), approximately 100,000 tonnes of lead are used annually in the manufacture of lead shot and ammunition. Lead stampings, pressings and castings are also still widely used for many weighting applications, including wheel balance weights, fishing weights, weights for analytical instruments and yacht keels (IARC 2006). Lead can also be present in castings used for soldering, stained glass articles, leaded glazes to make pottery, blown glass and screen-printing preparations (Grabo 1997). Furthermore, some traditional folk remedies can contain high levels of lead, but these uses are not permitted in Canada. Hispanic remedies taken for an upset stomach contain over 90% lead by weight of lead oxide in the case of Greta remedy, and lead tetraoxide in the case of Azarcon remedy (Baer and Ackerman 1988; United States Centers for Disease Control and Prevention [CDC] 2010). In addition to herbs, ashes and other materials, traditional kohl contains lead, principally lead sulphide but potentially also lead tetraoxide or lead carbonate (Alkhawajah 1992; HC 2010*c*; Vaishnav 2001). Food supplements and cookware containing lead (American Academy of Pediatrics Committee on Environmental Health 2005) and certain foods or beverages, such as illegal moonshine, have been associated with elevated BLLs (Dobrowolska and Melosik 2008; Hunt *et al.* 2009; Morgan and Parramore 2001). Vintage wines containing lead foil capsules or those made in wineries equipped with brass fittings can also present sources of sporadic and/or elevated lead exposure. Plastic food wrappers can further contribute to the dietary intake of lead (ATSDR 2007*a*).

The Children's Jewellery Regulations provides a guideline limit for lead in children's jewellery (Government of Canada 2018*a*). In addition, the Consumer Products Containing Lead Regulations

have been put in place to limit the total lead content in an expanded scope of consumer products, including products in contact with food and those used for sleep and hygiene (Government of Canada 2018*b*). There are currently no approved uses of lead in cosmetics or food in Canada.

Due to lead's ubiquitous nature, it may also be found in many products as unintended residues or impurities. In Canada, lead may be found as an impurity in a small number of pesticide products, several rodenticides, antifouling paints, and in the technical grade of several active ingredients (HC 2013*a*). However, the concentrations of lead in products derived from such sources are not assumed to represent a large source of exposure.

Following the virtual elimination of lead in household paints, gasoline additives and solder in food cans, the production of batteries comprises the single largest global market for refined lead today (75%) (Keating 1995; Keating and Wright 1994; NRCan 2010; OECD 1993).

2.7 Human Tissues and Biological Fluids

2.7.1 Organs

Most of the data on lead body burdens is derived from on autopsy samples from the 1960s and 1970s (Barry 1975; 1981; Gross *et al.* 1975; Schroeder and Tipton 1968). Reported tissue and organ concentrations are relatively constant in adults (Barry 1975; Treble and Thompson 1997), reflecting the faster turnover of lead in soft tissue relative to bone. Schroeder and Tipton (1968) reported relative lead in soft tissue: liver, 33%; skeletal muscle, 18%; skin, 16%; dense connective tissue, 11%; fat, 6.4%; kidney, 4%; lung, 4%; aorta, 2% and brain, 2% (other tissues were < 1%). Later work (Barry 1975; Gerhardsson *et al*. 1986; 1995; Gross *et al*. 1975; Oldereid *et al*. 1993) confirmed the liver as the soft tissue containing the highest lead concentrations in adults, followed by the kidney.

2.7.2 Blood

Canadian BLLs have been declining steadily since 1978–1979, when the geometric mean BLL was 4.79 µg/dL among people aged 6 to 79 years (Bushnik *et al.* 2010; HC 2019*a*). This decline is attributed to the successful phase-out of lead in gasoline, lead-based paints and lead solder in food cans, in addition to other government regulation and industry action over this time period (Bushnick *et al*. 2010; CDC 1991; HC 2013*a*; Thomas *et al*. 1999; Wang *et al*. 1997). Studies in northern populations (Nunavut and Northern Québec), indicated that house dust (Fillion *et al*. 2014) and ammunition (Fillion *et al*. 2014; Lévesque *et al*. 2003) were the main sources of lead for that population and that soil and food concentrations were low.

HC indicates that the average BLL for 3- to 79-year-olds has decreased over the period of the Canadian Health Measures Survey program, from 1.66 µg/dL for the 2007 to 2009 period in 6- to 79-year-olds (HC 2010*d*) to 0.93 µg/dL for the 2016 to 2017 period in 3- to 79-year-olds (HC 2019*c*) (see Appendix 1). The First Nations Biomonitoriing Intitiative measured blood from 13 First Nations communities in northern Canada in 2011 and found BLLs of 1.15 and 3.27 µg/dL (GM and 95th percentile, respectively) (AFN 2013). Analyses of 2017–2018 National Health and

Nutrition Examination Survey (NHANES) data for the United States (CDC 2021) found BLLs similar to, but slightly higher than the Canadian results: the geometric mean (for 1- to 5-year-old children were 0.670 µg/dL (95% confidence interval [CI] 0.600, 0.748), and 0.855 µg/dL (0.816, 0.895) in adults ≥ 20 years of age.

Under most exposure scenarios, BLLs show a characteristic age trend (CDC 2004; Statistics Canada 2013). From birth up until 6 months of age, an infant's BLL will reflect that of its mother (Schell *et al*. 2003) due to transfer through the placenta or breast milk (Manton *et al.* 2000). Around 6 months of age, environmental lead exposures (e.g., house dust, lead-based paint, soil, products) gradually increase through hand-to-mouth behaviour and the mouthing of non-food objects (Manton *et al*. 2000; Tulve *et al*. 2002). A peak in BLL in toddlers (1–3 years) was confirmed by several longitudinal studies of lead-exposed children (Baghurst *et al.* 1987; Canfield *et al.* 2003*a*; Dietrich *et al*. 1993; 2001; Zahran *et al*. 2011), while other studies show a subsequent decline in BLLs from the age of 3 onwards (Bell *et al*. 2011; Institut national de santé publique du Québec [INSPQ] 2011; Richardson *et al*. 2011; Statistics Canada 2013). After BLLs show a slight decreasing trend during childhood and adolescence, they rise again with age. Seniors typically have the highest BLLs, which result from exposure to higher environmental lead concentrations in the past and the remobilization of accumulated bone lead into the blood stream (HC 2013*a*).

2.7.3 Bone

The molecular characteristics of lead (Pb^{2+}) are very similar to those of calcium (Ca^{2+}), and lead preferentially accumulates in bones, where it has an elimination half-life of years to decades. Bone represents about 90% of the total lead body burden for adults and about 70% for children (Barry 1975). Lead preferentially accumulates in the bone regions undergoing the most active calcification at the time of exposure (Aufderheide and Wittmers 1992; US EPA 2006*b*) and bone lead (PbBO) concentrations increase with age, indicating a relatively slow turnover rate for lead in adult bones (Barry 1975; 1981; Gross *et al*. 1975; Schroeder and Tipton 1968).

Lead in bone is in continuous exchange with soft tissue, such as blood, and can contribute to elevated BLLs even after external exposures are eliminated (Fleming *et al*. 1997; Inskip *et al*. 1996; Kehoe 1987; O'Flaherty *et al*. 1982; Smith *et al*. 1996). Additionally, physiological states of stress such as pregnancy, lactation, menopause/andropause, extended bed rest, hyperparathyroidism and osteoporosis are all associated with increased bone resorption and subsequent increases in BLLs (Franklin *et al.* 1997; Gulson *et al.* 1997*b*; 1999*b*; 2003; Silbergeld *et al*. 1988).

2.7.4 Other Tissues and Biological Fluids

Other body tissues and fluids such as urine, feces, teeth, nails, hair, sweat, saliva and breast milk also sequester absorbed lead (Chamberlain *et al.* 1978; Griffin *et al.* 1975; Hursh and Suomela 1968; Hursh *et al.* 1969; Kehoe 1987; Rabinowitz *et al.* 1976; Stauber *et al*. 1994). Some data on tissue concentrations are available in the Canadian Health Measures Survey (HC 2019*c*).

3. ENVIRONMENTAL FATE AND BEHAVIOUR

Lead is a naturally occurring metal that enters the environment from a variety of sources. Following industrialization, anthropogenic emissions have become responsible for the ubiquitous presence of lead in the atmosphere, water, sediment, soils and biota (CDC 1991).

The primary factors affecting the migration of lead from particulate matter deposits to the biosphere include groundwater leaching and physical-chemical conditions (Förstner 1987). 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 lead mobility include pH, hydroxides, oxyhydroxides, oxides, clay minerals, organic matter and complexing anions. These properties affect the adsorption, precipitation and co-precipitation of lead. Physical properties affecting lead dispersion include the texture of sediment and soil (grain size and surface area), biota characteristics (type, organ, age, root distribution) and seasonal changes.

The following discussion is intended to provide a brief overview of the fate and persistence of lead in the environment.

3.1 Atmosphere

The atmosphere is the major initial recipient of lead, with anthropogenic sources contributing at least 1–2 orders of magnitude more lead than natural sources (Komárek *et al*. 2008). Lead emissions from low-temperature processes tend to exist as larger particles and rapidly settle, whereas high-temperature sources (forest fires, volcanic eruptions and anthropogenic releases) produce smaller particles with longer residence times (Hill 1992; Jaworski *et al.* 1987). Atmospheric lead sourced from natural and anthropogenic sources is found as both gaseous (volatile organo-lead compounds, generated predominantly from leaded-gasoline combustion) and particulate matter (Hill 1992). Partitioning is skewed toward the particulate phase since volatilized lead rapidly condenses or adheres to aerosols and other airborne particulates (EC 1994; Häsänen *et al.* 1986). In the atmosphere, organic lead compounds are dealkylated to ionic trialkyl lead, dialkyl lead and eventually into inorganic lead (Harrison and Laxen 1978; Jarvie *et al.* 1981) through a combination of direct photolysis and reactions with hydroxyl radicals and ozone (ATSDR 2007*a*). Therefore, most atmospheric lead is in the form of inorganic species, predominantly sulphates, halides and oxides (Kabata-Pendias 2001; Kabata-Pendias and Pendias 1992; Pacyna 1987). Lead halides can further be converted to lead oxides, carbonates and sulphates (Davies 1995; Kabata-Pendias 2001).

Lead atmospheric residence times vary from several hours to several days (Jaworski *et al.* 1987). ATSDR (2007*a*) estimated an average 10-day residence time, while OECD (1993) estimated seven- to 24-day residency for lead from vehicular exhaust, and Pilgrim (1995) calculated a typical residence time of 2.5 days for lead from the Belledune, New Brunswick, smelter.

Deposition rates are greatest near the source of emissions and decrease with distance (Weis and Barclay 1985), usually in an exponential fashion (Wixson and Davies 1993). More lead is removed

from the atmosphere through wet deposition than dry deposition. The scavenging capacity of wet deposition is a function of local rain chemistry and quantity; for example, greater amounts of lead are expected to be removed under low-pH conditions due to more lead being dissolved (OMEE 1994).

Lead deposition from automobile exhaust was generally reported to occur within 100 m of the roadway (RSC 1986). However, owing to the long residence time of small particle-bound lead, long-range transport (hundreds and thousands of kilometres) has been reported (Haygarth and Jones 1992; Pilgrim 1995), resulting in increased deposition in remote areas (Ewers and Schipköter 1991).

3.2 Surface Water, Groundwater and Sediment

Once deposited in water, lead rapidly partitions among three phases: dissolved, suspended particulate and sediment lead (Mackay and Diamond 1989). In aquatic environments, most lead mass is associated with sediment (OMEE 1994), with the remainder distributed between suspended particulates and dissolved in water (filtrate) at ratios generally ranging from 4:1 in rural streams to 27:1 in urban streams, depending on the amount of suspended matter (US EPA 2006*a*). The capacity of sediment to hold lead depends on the organic and oxyhydroxide content as well as particle size (OMEE 1994). Microbial methylation of lead can occur in sediments (Andreae and Froelich 1984; Beijer and Jernelöv 1984; Hewitt and Harrison 1987; Jarvie 1988; Walton *et al.* 1988).

Freshwater lead chemistry is complex, as lead exists in a multiplicity of forms of varying solubility and environmental mobility (ATSDR 2007*a*). These forms include: 1) the mobile and available free and solvated Pb^{2+} ions, ion pairs and ion polymers such as $Pb(OH)_2$ and $Pb(OH)_3^-$ (Hill 1992); 2) the slightly soluble and insoluble compounds such as PbO, $PbCO_3$, and $PbSO_4$ (Hill 1992) formed with the major anions found in natural waters (ATSDR 2007*a*); 3) the strongly bound and mostly biologically unavailable organic lead complexes formed with dissolved humic materials (CCREM 1987); 4) the strongly bound, yet somewhat mobile and available lead attached to colloidal particles such as iron oxide; and 5) the mostly unavailable lead of sparingly soluble lead hydroxide precipitates (pH 5–7) (CCREM 1987) and lead sorbed onto solid particles of clay or the remains of organisms (OECD 1993).

The speciation of lead in water is related to solubility (CCREM 1987), which is affected by pH, redox potential and the presence of sulphide, sulphate and carbonate ions (ATSDR 2007*a*). Its water solubility is governed by pH and the presence of dissolved salts (ATSDR 2007*a*). Lead sulphide, although insoluble, may be slowly oxidised to create the more soluble sulphate. At pH values < 5.4, lead sulphate acts as a solubility control, whereas at pH values > 5.4, lead carbonates act as the control. Between a pH of 6 and 8, lead solubility is a complex function dependent on pH and carbonate; at constant pH, increased solubility occurs as alkalinity decreases (CCREM 1987). As pH decreases, lead solubility increases, with Pb^{2+}, in the form of free or solvated ions and ion pairs, rising about two orders of magnitude with each pH unit drop (EC 1994). Solubility can also be altered by water temperature (OMEE 1994), as well as organic, metal salt and suspended sediment content (CCREM 1987; OMEE 1994).

3.3 Soil

Natural levels of lead in soil reflect the mineralogy of the soil's parent (geological) material. Lead is present in soil as the soluble plumbous ion (Pb^{+2}), in precipitated forms as carbonates, sulphates and oxides, and in the soil lattices as lead silicates (Davies 1995; Kabata-Pendias 2001). Reports suggest lead is mainly adsorbed onto clay minerals, adsorbed and co-precipitated with manganese oxides and iron and aluminum hydroxides, adsorbed onto colloidal organic matter, and complexed with organic moieties.

Lead is persistent in soil because of its low solubility (Carelli *et al.* 1995; Leita and De Nobili 1991), its strong complexing behaviour with organic matter (Kabata-Pendias 2001), and its relative freedom from microbial degradation (Davies 1995; OECD 1993). Due to lead's relatively low leaching potential, soils and sediments are generally considered to be mass sinks (Davies 1995; Hill 1992; OECD 1993; Stokes 1989). The soil's pH and its levels of humic acid, fulvic acid, clay and organic matter content can influence the potential for leaching (EC 1994; Ewers and Schipköter 1991). Acidic conditions favour lead solubilisation, and acidic soils tend to have lower lead contents (Kabata-Pendias 2001). Nelson and Campbell (1991) suggested humic and fulvic acid interactions, rather than acidification, may be the primary mechanism through which lead leaching occurs. Lead is not degraded in the environment, although some fate processes can transform certain lead species into others (ATSDR 2007*a*; Carelli *et al.* 1995). The biomethylation of inorganic lead can occur, but it is not considered a significant mobilization process (Andreae and Froelich 1984; ATSDR 2007*a*; Beijer and Jernelöv 1984; CCREM 1987; Walton *et al.* 1988). Organic lead forms are more likely to undergo microbial-mediated reactions (ATSDR 2007*a*).

The prevailing forms of lead in soil associated with high-temperature combustion processes, such as smelter operations, include sulphide (PbS), the sulphates ($PbSO_4$ and $PbO.PbSO_4$), and the oxides (PbO and PbO_2) (Hemphill *et al.* 1991).

3.4 Indoor Dust

Indoor dust contains approximately five times the organic matter of soil (Rasmussen *et al.* 2008), and some metals, including lead, have an affinity for this organic matter (Rasmussen 2004). Decreasing particle size was also correlated with increased lead content, due to the increased surface area-to-mass ratio (Rasmussen *et al*. 2008).

Lead carbonate is the dominant species in house dust, comprising 17–76% of total lead (MacLean *et al*. 2011). The portion of house dust lead that is bioaccessible was moderately correlated with the age of homes, although 10% of the homes with higher levels of bioaccessible dust lead (geomean = 1,730 mg/kg soluble lead) were built after 1980 (McDonald *et al*. 2010; 2011; Rasmussen *et al*. 2011). The authors indicate that approximately 90% of urban Canadian homes have bioaccessible dust lead levels < 250 mg/kg. Bioaccessible lead was shown to include > 50% highly bioaccessible forms (84–92% relative bioavailability) (MacLean *et al*. 2011): organic lead citrate and lead absorbed to iron- or aluminum-oxyhydroxides, inorganic lead carbonate or lead hydroxyl carbonate (hydrocerussite) (Rasmussen *et al*. 2011).

4. BEHAVIOUR AND EFFECTS IN BIOTA

The assessment of behaviour and effects in biota has not been updated in this update of the $PSoQGs_{HH}$. Some more recent data on average levels of lead in North American foods are presented in Appendix 1.

5. BEHAVIOUR AND EFFECTS IN HUMANS AND MAMMALIAN SPECIES

5.1 Overview

With respect to its toxicity, lead is one of the most extensively studied substances (Needleman and Gatsonis 1990). Lead has no known biological function (White *et al*. 2007).

As discussed previously in Section 2.0, lead may be found in a variety of natural and anthropogenically derived chemical forms. Measurements of lead in environmental and biological media rarely identify the form, but rather refer to the lead moiety contained within unspecified substances; therefore, for the purposes of this toxicological review, specific substances are explicitly identified, and "lead" refers to the Pb^{2+} moiety. Nevertheless, consideration of these other forms may be warranted at sites where other forms of lead are expected to predominate.

Lead is widely distributed throughout the body. Overall, the mode of action for lead is attributed to its affinity for thiol groups (-SH) and other organic ligands in proteins (Vallée and Ulmer 1972), which are common to all cell types (ATSDR 2020). Furthermore, several authors consider that lead's ability to substitute for calcium (and perhaps other essential metals such as zinc) factors into its mode of action (ATSDR 2007*a*; Bressler and Goldstein 1991; EFSA 2013; Hanas *et al*. 1999; Zawia *et al*. 1998). Like calcium, lead accumulates in bones and can cross cellular membranes through the voltage-regulated Ca^{2+} channel (Calderon-Salinas *et al*. 1999) or via active transport systems such as Ca^{2+}-Mg^{2+}-ATPase (Simons 1988).

5.2 Classification

Both cancer and non-cancer endpoints have been identified in the toxicological database for lead. The International Agency for Research on Cancer classifies lead as Group 2A, probably carcinogenic to humans, based on occupational inhalation data and *in vivo* oral studies (IARC 2006). The National Toxicology Program concludes that Pb and Pb compounds are reasonably anticipated to be human carcinogens (NTP 2016). The weight of evidence supports the conclusion that soluble inorganic lead is a carcinogen in animal experiments and that the kidney is the most sensitive site of tumour occurrence in lead-exposed rodents. Evidence also suggests that lead promotes renal tumours in rats, and epidemiological evidence indicates that lead is likely to be carcinogenic at high doses (HC 2019*a*).

5.3 Toxicokinetics

5.3.1 Absorption (Bioavailability)

On the molecular level, lead is very similar to calcium. Therefore, once in the body, its kinetics are governed by calcium dynamics and consequently, any inorganic lead present in the body is distributed in essentially the same manner, regardless of the route of exposure (Chamberlain *et al.* 1978; Kehoe 1987).

Lead can enter the body through various routes of exposure and environmental media. The rate and extent of absorption via each medium is defined as bioavailability. The bioavailability of lead can be expressed either in absolute terms (absolute bioavailability) or in relative terms (relative bioavailability). Absolute bioavailability can be defined as "the fraction or percentage of a compound which is ingested, inhaled or applied on the skin surface that is absorbed and reaches the systemic circulation" (Ruby *et al*. 1999; US EPA 2007*b*). Some bioavailability experiments involve an assessment of absorption from a target media (e.g., contaminated soil) relative to absorption of a highly soluble lead form, or compared with lead excretion or blood concentrations following intravenous administration (100% absorption). When absorption is measured relative to a reference material, it is referred to herein as relative bioavailability. More specifically, relative bioavailability can be defined as "a measure of the difference in extent of absorption among two or more forms of the same chemical (e.g., lead carbonate vs. lead acetate), different vehicles (e.g., food, soil, water) or different doses" (Roberts 2004; Ruby *et al*. 1999; US EPA 2007*b*).

Several key methods have been used to measure the bioavailability of lead:

1. lead dosing followed by measurement of lead in excreta and/or tissue concentrations in humans and animals
2. radioisotopic studies in humans
3. *in vitro* experiments using synthetic digestive fluids and soluble forms of lead to determine a bioaccessiblity value that is subsequently converted to a relative bioavailability value (see Section 5.3.1.2).

Bioaccessibility is a term used to describe the fraction of lead in a sample of a given exposure medium (e.g., soil) that has been dissolved in an *in vitro* system that mimics dissolution in the gut (most often in simulated or real gastric fluid) (Schoof 2003). Bioaccessibility cannot be used directly to estimate the amount absorbed into systemic circulation; it requires validation with the bioavailability of a soluble form of lead to determine the absorbed fraction. Bioaccessibility provides an estimate of relative bioavailability (Ruby *et al.* 1999; US EPA 2007*a*) and is a measure of physiological solubility (US EPA 2017).

Lead absorption in humans is influenced by various factors, including age and physiological status, chemical/physical state of lead compounds, food intake and nutrient status (particularly in relation to calcium and iron demands) (ATSDR 2007*a*; US EPA 1994*a*; US EPA 2006*b*). Pounds *et al.* (1978) found greater oral absorption for a highly soluble form of lead (lead acetate) in juvenile female Rhesus monkeys, compared to adult females. Similar results were found in rodents (Kostial *et al.* 1978). Nutritional iron and calcium deficiencies in children both appear to increase lead

absorption (EFSA 2013). Both active and passive mechanisms exist for lead absorption in the gut (US EPA 1994*a*) and calcium competes with lead uptake via the active mechanism (Fullmer *et al*. 1985). Some studies suggest there may be a genetic phenotype associated with increased lead absorption (a variant of the ALA-D gene; US EPA 2006*a*). Experimentally determining the accuracy of bioavailability estimates is further complicated by the contribution of endogenous bone sources to BLLs (Gulson *et al.* 1999*a*; Manton *et al.* 2000). The US EPA (2003) has estimated that the rate of lead absorption in adults is approximately 40% that of children. Overall, the relationship between BLLs of children and adults and the concentration of lead in water and in food appears to be curvilinear, and to be near-linear at low doses (HC 2013*a*). Bioavailability factors considered in the derivation of the $PSoQG_{HH}$ include the age of the exposed population, media of exposure, and nutrient/dietary status.

5.3.1.1 Oral Absorption from Water

The majority of data on the bioavailability of lead in water are derived from studies with adult humans. Fasting significantly increases the bioavailability of lead in water (67.4–75.8% [mean = 69.5%] in fasted vs. 11.2–23.3% non-fasted subjects [mean = 15.3%]) (Blake 1976; Blake *et al*. 1983). When minerals were added to distilled water to create "hard" water, bioavailability decreased to a range of 0.94 to 1.5% (Blake *et al*. 1983). Absorption of lead via the digestive tract also greatly increases when the dietary intakes of calcium and phosphorus are low (Blake and Mann 1983; Blake *et al*. 1983).

Lead is assumed to be more bioavailable in water than in food (US EPA 2007*b*). HC (2019*a*) completed additional pharmacokinetic modelling for lead in water, which resulted in an overall adjustment of the EFSA (2013) TRV of 0.5 µg/kg body weight per day (bw/d) when lead is found in food to an equivalent value of 0.4 µg/kg bw/d when lead is in water.

5.3.1.2 Absorption from Soil or Dust

In general, gastrointestinal absorption (bioavailability) of lead in either soil or house dust is governed by three primary factors: 1) soil or solid medium characteristics (e.g., organic carbon and clay content); 2) chemical form (e.g., metal compounds or minerals or mixtures of organic chemicals) and the nature of their interactions with soil particles; and 3) the nature of soil contact by the receptor of interest (e.g., hand-to-mouth contact by a child) (National Research Council 2003; Ruby *et al*. 1999; Scheckel *et al*. 2009).

Data from bioavailability studies using rodents, swine and humans demonstrate that gastrointestinal absorption varies with the age, diet, and nutritional status of the subject, as well as by chemical species and particle size (Bradham *et al*. 2014; HC 2017*a*). Lead bioavailability has been more extensively studied in soil. Furthermore, soil physical and chemical characteristics such as texture, clay mineralogy, organic carbon content, cation exchange capacity (CEC), redox potential, and pH, along with the presence of other elements (e.g., sulphur, nitrogen, phosphorus, etc.) will influence lead's solubility and mobility in the environment and its accessibility in the gastrointestinal tract (Datta and Sarkar 2005; Hack and Selenka 1996; Kabata-Pendias 2001; Kelley *et al.* 2002; National Research Council 2003; OECD 1993; Panis and Lucianer 1987; Ruby

2004; Yang *et al.* 2002). Using a juvenile swine model and soil samples primarily from lead mining/smelting sites in the United States, a National Institute of Standards and Technology (NIST) paint sample- (comprised primarily of fine particles < 5 µm) and a galena-enriched soil, Casteel *et al.* (2006) estimated relative bioavailability for soil, grouped by mineralogy, at 0 to 99.2% over a soil lead concentration range of 1,270 to 14,200 ppm (summarized in Appendix 2). Based on this study and previous work performed by Casteel *et al.* (1998), the US EPA has provided a default bioavailability adjustment and detailed guidance for estimating the relative bioavailability of lead in soil and soil-like materials using either a swine model or an *in vitro* model (US EPA 2017).

Soil (and dust) organic carbon content affects lead bioavailability. These levels tend to be higher in dust than in exterior soils (Rasmussen 2004; Rasmussen *et al.* 2001) (see section 2.5.3) and have consistently higher migratable concentrations of lead, as determined using simulated stomach acid digestion (Rasmussen 2004; Rasmussen *et al*. 2008).

Maddaloni *et al*. (1998) estimated the absolute bioavailability of lead in fasted adult volunteers at 26.2%. Within the blood compartment, 14.4% of the administered dose was detected 24 hours post-exposure, but only 1.4% and < 3% of the administered dose was detectable within the blood compartment in those who ate a high-fat or standardized breakfast, respectively, prior to dosing. For children, von Lindern *et al.* (2003) predicted a soil/dust bioavailability of 18% (range: 12–23%) based on regression with BLLs and varying soil concentrations from an exposed cohort.

Various *in vitro* bioaccessibility tests exist to estimate the oral bioavailability of contaminants in soil and dust. Significant variability in lead bioavailability from soil and dust has been estimated in several studies using these tests (Dodd *et al*. 2014; 2017; Drexler and Brattin 2007; Intrinsik Inc. 2010; Rasmussen *et al*. 2014; Royal Roads University 2002; Sudbury Area Risk Assessment 2008; Turner and Ip 2007; US EPA 2017; Yu *et al*. 2006). Given the wide variability in bioavailability, as assessed in Canadian soils (Dodd *et al.* 2017), CCME assumed a relative bioavailability of 1 for the $PSoQG_{HH}$.

5.3.1.3 Absorption from Food

Similarly to absorption from water, gastrointestinal absorption was highest in subjects who had fasted prior to ingesting lead (Blake *et al*. 1983; James *et al*. 1985; Rabinowitz *et al*. 1980). Bioavailablity was variable across studies (Alexander 1974; Blake 1976; Blake *et al.* 1983; Gulson *et al.* 1997*b*; James *et al.* 1985; Manton *et al.* 2000; Rabinowitz *et al.* 1976; 1980; Ziegler *et al.* 1978). However, bioavailability was lower (1–5%, theoretical maximum of 12%) in young children (< 8 years) in one study (Manton *et al*. 2000) than in older children (10–15% in children 6–11 years old) (Gulson *et al.* 1997*b*).

Bioavailability studies are summarized in HC (2019*a*).

5.3.1.4 Absorption via Inhalation

The bioavailability of inhaled lead has been studied primarily in adult humans exposed to fine particulate–borne lead. Pulmonary absorption depends on the size (US EPA 2006*a*), lead solubility and breathing patterns (i.e., number of breaths per minute) of the exposed individual (ATSDR 2007*a*). Organic forms of lead may be more readily absorbed through the lungs. Heard *et al.* (1979) suggested bioavailability may be as high as 80% for organic forms of lead.

Clearance of the upper respiratory tract results in the swallowing and gastrointestinal absorption of the larger lead particles. Lead particles smaller than < 1 µm in diameter and lead fumes reach the lower respiratory tract and may be absorbed through extracellular dissolution or through ingestion by phagocytic cells (ATSDR 2007*a*). Therefore, it is important to characterize the proportion of lead deposited in the lungs in addition to the absorption of lead through the lining of the lungs and into blood.

Available data suggest nearly complete absorption (95 – ≈100%) of the particulate-bound lead that reaches the lungs (25% from < 1 µm size fraction and 35% from automotive exhaust) (Chamberlain *et al.* 1975; Klaassen 1996; Morrow *et al.* 1980; Wells *et al.* 1975). The US EPA (1994*b*) estimated that 25 to 45% of particles were deposited in the lungs of children. A relative absorption factor (RAF) of 1 was assumed for the purpose of $PSoQG_{HH}$ derivation for inhalation exposures.

5.3.1.5 Absorption through the Dermis

The following relative (to inhalation) dermal absorptions for five forms of lead were determined from human and rat dermal exposures: lead naphthalene (0.170%), lead nitrate (0.030%), lead stearate (0.006%), lead sulphate (0.006%), lead oxide (0.005%), and lead metal powder (0.002%) (ATSDR 2007*a*; Bress and Bidanset 1991; Sun *et al.* 2002). Studies of penetration rates through excised human and guinea pig skin suggest organic lead forms are absorbed to a greater extent than inorganic forms: tetra butyl lead > lead nuolate (lead linoleic and oleic acid complex) > lead naphthanate > lead acetate > lead oxide (nondetectable) (Bress and Bidanset 1991).

Based on a study with ^{203}Pb as lead acetate in cosmetics in male human volunteers (n = 8) under various treatment groups (Moore *et al.* 1980), the greatest dermal absorption was approximately 0.3% (after dry application to scratched skin) (other groups had ranges of mean values of 0.04 to 0.2%). Using the absolute dermal absorption value of 0.3% and an assumption of 50% oral absorption from food and water, CCME estimated a dermal RAF of 0.006 (i.e., 0.3%/50% = 0.006) and used this RAF to estimate the $PSoGQ_{HH}$.

5.3.2 Distribution

The distribution of absorbed lead can be divided into at least two major kinetic pools with differing turnover rates: soft tissues (e.g., blood) and the skeleton (i.e., bone). The skeleton acts as a reservoir for lead with a biological half-life of about 17 to 27 years (ATSDR 2007*a*; Rabinowitz *et al*. 1976), substantially longer than the half-life for blood and other soft tissues (16 to 40 days) (Chamberlain

et al. 1975; Gulson *et al.* 1996; Rabinowitz *et al*. 1976). Approximately 90% of the total adult lead body burden and ≈ 70% of the total in children is present in bone (Barry 1975). Published data suggest that the uptake and release of lead from the skeleton has a significant effect on BLLs.

Distribution in various compartments is summarized below. More details are also available in HC (2019*a*).

5.3.2.1 Lead in Soft Tissues and Organs

Under steady-state conditions, 96 to 99% of lead in the bloodstream is bound to cellular proteins within red blood cells and is thus not readily available to cross into other tissues or organ systems (ATSDR 2007*a*; Bergdahl *et al*. 1997; 1998; 1999; Goyer 1990; Hernandez-Avila *et al*. 1998; Manton *et al*. 2001; Schütz *et al*. 1996; Smith *et al*. 2002). Of the remaining lead in the bloodstream, ≈ 40–70% is bound to plasma proteins, particularly albumin, while unbound serum lead exists primarily as low-molecular-weight sulfhydryl compounds such as cysteine and homocysteine (Al-Modhefer *et al.* 1991; Ong and Lee 1980). While plasma contains only a minor fraction of blood lead, it can easily exchange with soft tissues and thus may be the more toxicologically active fraction of circulating lead (Ambrose *et al*. 2000; Campbell *et al*. 1984; Skerfving *et al*. 1993). Circulating blood lead represents less than 1% of the total body burden, while soft tissues and organs represent about 8% (EFSA 2013).

Blood lead can cross cell membranes via anion exchange and the calcium-ATPase pump transport mechanisms or via calcium channels. While lead is distributed to many organs and soft tissues, the liver and kidney cortex have the highest soft-tissue concentrations in adults (Barry 1975; Gerhardsson *et al*. 1986; 1995; Gross *et al*. 1975; Oldereid *et al*. 1993).

5.3.2.2 Lead in Bone

Bone can be divided into two generally distinct types: compact cortical bone and cancellous trabecular bone (Ambrose *et al.* 2000). Because of its larger surface area, trabecular bone has a higher rate of turnover than cortical bone, and lead in trabecular bone is more labile than in cortical bone (Ambrose *et al*. 2000; Klaassen 2008) and may have more influence on BLLs (Hu *et al*. 1989). As lead preferentially accumulates in regions undergoing the most active calcification, it is thought to accumulate predominantly in trabecular bone during childhood and in both trabecular and cortical bone during adulthood (Aufderheide and Wittmers 1992). Dowd *et al*. (2001) demonstrated *in vitro* that lead can covalently bind to osteocalcin in the mineral matrix with a greater strength than calcium.

Bone formation exceeds the rate of bone resorption during the first two decades of life (Ilich and Kerstetter 2000), which can lead to a net accumulation of lead in the bone. Peak bone mass occurs at ≈ 30–35 years of age (O'Flaherty 2000). Bone resorption and a net decrease in bone mass can exceed bone formation at older ages, as well as during pregnancy and menopause/andropause (Gulson *et al.* 2002; 2003; 2004; Hernandez-Avila *et al.* 2000; Symanski and Hertz-Picciotto 1995).

Resorption of bone is mediated by osteoclasts, which release bone matrix and associated calcium and lead back into the bloodstream (Berglund *et al.* 2000; Sharan *et al.* 2008). Bone biokinetics can be influenced by calcium levels (Berglund *et al.* 2000; Sharan *et al.* 2008); pregnancy (Markowitz and Shen 2001; Naylor *et al.* 2000); levels of micronutrients, such as magnesium and phosphorous, in bone (Anast *et al*. 1976; Gulson *et al.* 2006); estrogen levels (Khosla 2001; Nasu *et al.* 2000); and levels of other hormones, such as calcitonin, androgen, and growth factors (Berglund *et al.* 2000). Bone biokinetics are a key determinant of BLLs and significant correlations have been observed between BLLs and bone lead (PbBO) in older adults (Chettle 2005; Fleming *et al.* 1997), peri-menopausal and pregnant women (Markowitz and Shen 2001; Naylor *et al.* 2000; O'Flaherty 2000; Popovic *et al.* 2005), and children (Gulson *et al.* 1997*b*; Gwiazda *et al.* 2005; Manton *et al.* 2000; McNeill *et al.* 2000), with more or less significant contributions depending on relative bone formation to bone resorption rates.

Several studies have found stronger correlations between PbBO and adverse outcomes than BLLs and adverse outcomes in adults (Cheng *et al.* 2001; Glenn *et al.* 2003; Korrick *et al.* 1999; Rothenberg *et al.* 2002*a*).

5.3.2.3 Transfer to Fetus/Infant during Pregnancy and Breastfeeding

Lead is readily transferred from the mother to the fetus *in utero* and to infants via breast milk (HC 2013*a*). Gulson *et al.* (2004) determined that BLLs are elevated during pregnancy due to endogenous sources and increased by an average of 65% (range: 30–95%) postpartum compared to late pregnancy. Lead levels in breast milk and maternal BLLs are significantly correlated with infant BLLs (Carbone *et al.* 1998; Ettinger *et al*. 2004; Graziano *et al.* 1990; Gulson *et al.* 1997*a*; 2003; Koyashiki *et al*. 2010). Researchers estimate that 79–90% of the mobilized lead in pregnant women can reach the fetus via cord blood (Gulson *et al*. 2003; Mahaffey 1991). Lead is primarily concentrated in fetal bone (Mahaffey 1991) and may also be detected in soft tissues, including the liver, heart and brain.

5.3.3 Metabolism in Humans

5.3.3.1 Organic Lead

Active metabolism of alkyl lead compounds occurs through oxidative dealkylation by CYP-450 in the liver. The metabolic pathways and products for organic lead compounds have not been fully elucidated, although occupational monitoring studies indicate workers occupationally exposed to tetraethyl lead excrete diethyl lead, ethyl lead, and inorganic lead in their urine (Turlakiewicz and Chmielnicka 1985; Vural and Duydu 1995; Zhang *et al*. 1994). Trialkyl lead metabolites have also been identified in brain tissue of non-occupationally exposed individuals and in the liver, kidney and brain tissue of workers (Bolanowska *et al.* 1967; Nielsen *et al.* 1978).

5.3.3.2 Inorganic Lead

The metabolism of inorganic lead primarily involves the formation of complexes with various proteins and nonprotein ligands (ATSDR 2007*a*; US EPA 2006*a*). To date, major extracellular ligands include albumin and nonprotein sulfhydryls. Within red blood cells, the major intracellular ligand is ALA-D, while in the plasma approximately 40 to 70% of lead is bound to albumin. Unbound serum lead exists primarily as low-molecular-weight sulfhydryl compounds, such as cysteine and homocysteine (Al-Modhefer *et al.* 1991; Ong and Lee 1980).

In other soft tissues and organs, lead is predominantly bound to proteins. Results of *in vivo* rodent studies identified high-affinity cytosolic lead-binding proteins (PbBPs) in male (but not female) rat kidneys and brains (DuVal and Fowler 1989; Fowler 1989). Specifically, these PbBPs were cleavage products of α_2-microglobulin, a member of the retinol-binding proteins superfamily (Fowler and DuVal 1991). Smith *et al*. (1998) also identified two high-affinity PbBPs (thymosin β_4 ($T\beta_4$, 5 kDa) and acyl-CoA binding protein (ACBP, 9 kDa) in the kidneys of environmentally exposed humans. *In vivo* studies conducted with rats have also demonstrated lead bonding to metallothionein, although it is not a significant inducer of the protein in comparison to cadmium and zinc (Eaton *et al*. 1980; Waalkes and Klaassen 1985).

5.3.4 Elimination

Lead is predominantly excreted via the urine (75%) and, to a lesser extent (25%), by biliary excretion into the feces (Klaassen 2008). Lead that is not absorbed via the digestive tract, along with the fraction of biliary excretion escaping entero-hepatic recirculation, is excreted in the feces (EFSA 2013). Sweat, saliva, hair, nails, and breast milk have also been identified as minor routes of excretion (Chamberlain *et al.* 1978; Griffin *et al.* 1975; Hursh and Suomela 1968; Hursh *et al.* 1969; Kehoe 1987; Rabinowitz *et al.* 1976; Stauber *et al*. 1994).

Overall elimination half-lives for lead in blood (30 days) and bone (27 years) have been identified (ATSDR 2007*a*), although the rate of excretion appears to vary with lead body burden. Reduced glomerular filtration was identified in individuals with relatively high body burdens, suggesting potential lead-induced renal toxicity (Yu *et al.* 2004).

5.4 Acute Toxicity

In humans, overt signs of acute lead intoxication include dullness, restlessness, irritability, poor attention span, headaches, muscle tremors, hallucinations and memory loss (ATSDR 2007*a*). Encephalopathy has been reported at relatively high BLLs (100–120 µg/dL in adults and 80–100 µg/dL in children) (NAS 1972; Smith *et al.* 1938). In adults, acute high-dose lead-induced nephrotoxicity is limited to functional and morphological changes in proximal tubular cells related to the formation of intranuclear inclusion bodies (Choie and Richter 1972; Goyer *et al.* 1970*a*; *b*) and mitochondrial abnormalities (Fowler *et al.* 1980; Goyer 1968; Goyer and Krall, 1969). Clinical manifestations of nephrotoxicity (aminoaciduria, glycosuria and hyperphosphaturia) have been observed following severe acute childhood exposures (Chisolm *et al*. 1955; Goyer *et al.* 1972;

Loghman-Adham 1998). Colic is also a known early symptom of lead poisoning following acute exposure to high levels of lead (at BLLs as low as 40 to 60 µg/dL) (ATSDR 2007*a*).

Anemia is another classic symptom of both acute and chronic lead poisoning (Landrigan and Todd 1994; WHO 2010). Lead-induced anemia resulting from an acute dose is caused by hemolysis, while chronic exposure leads to reduced erythrocyte lifespan and impaired heme synthesis.

ATSDR (2007*a*), US EPA (2006*a*), and EFSA (2013) all contain summaries of the acute effects of lead in laboratory animals and humans, and readers are referred to these publications for a broader overview of the health effects of acute exposure to lead.

5.5 Sub-Chronic and Chronic Systemic Toxicity

The sub-chronic and chronic effects of lead and its mechanisms of toxicity are well described in the literature, and several summaries of the epidemiological database for inorganic lead are publicly accessible (ATSDR 2007*a*; International Lead Zinc Research Organization and EBRC Consulting GmbH [ILZRO and EBRC] 2008; US EPA 2006*a*). Overall, evidence suggests that chronic lead exposure can manifest in myriad adverse effects including, but not limited to, neurotoxicological, cardiovascular, renal, endocrine, gastrointestinal, hematological, musculoskeletal, and reproductive effects. Most of these toxicological endpoints in humans have also been studied in experimental animals and the resulting *in vivo* data provide evidence to support the mechanisms of lead toxicity (EFSA 2013). Comprehensive reviews of mammalian lead toxicology have been published, including ATSDR (2007*a*); IARC (2006); EFSA (2013) and US EPA (2006*a*), and readers are referred to these publications for general overviews of the current toxicological database and the HC's Human Health State of the Science Report on Lead (HC 2013*a*). The summarized text on sub-chronic and chronic systemic effects draws heavily on the comprehensive toxicology review compiled by Healey *et al.* (2010). Sub-chronic and chronic toxicity studies are briefly described in this section.

5.5.1 Neurological Effects

Several lines of evidence demonstrate that both the central and peripheral nervous systems are principal targets for lead toxicity, and that neurological effects may occur at exposures associated with BLLs of < 10 µg/dL, including:

- neuromotor function (Boucher *et al*. 2012; Després *et al*. 2005; Dietrich *et al*. 1993; Fraser *et al*. 2006; Ris *et al*. 2004; Wasserman *et al*. 2000*b*)
- reduced academic achievement and reading or math skills (Al-Saleh *et al*. 2001; Chandramouli *et al*. 2009; Fergusson *et al*. 1997; Lanphear *et al*. 2000; Miranda *et al*. 2007; Needleman and Gatsonis 1990; Wang *et al*. 2002)
- delinquent or antisocial behaviour (Bellinger *et al*. 1994*b*; Dietrich *et al*. 2001; Fergusson *et al*. 1993; Needleman *et al*. 1996; 2002)
- impaired attention and executive function (Bellinger *et al*. 1994*a*; Bouchard *et al*. 2009; Braun *et al*. 2006; Canfield *et al*. 2003*b*; Chiodo *et al*. 2004; 2007; Cho *et al.* 2010;

Froehlich *et al*. 2009; Ha *et al*. 2009; Kim *et al.* 2010; Nicolescu *et al.* 2010; Nigg *et al*. 2008; 2010; Ris *et al*. 2004; Wang *et al*. 2008)

- impaired auditory function (Dietrich *et al*. 1992; Osman *et al*. 1999; Schwartz and Otto 1991)
- impaired visual function (Fox and Boyes 2008; Fox *et al*. 1997; Laughlin *et al*. 2008; Rothenberg *et al*. 2002*b*)
- age-related cognitive decline (Khalil *et al*. 2009; Shih *et al*. 2006; Stewart and Schwartz 2007; Weuve *et al*. 2009; Wright *et al*. 2003).

BLLs, as biomarkers of lead exposure, have also been associated with increased incidence of attention deficit hyperactivity disorder (ADHD) and behavioural problems (Aguiar *et al*. 2010). In 2012, NTP concluded that there is sufficient evidence that BLLs < 5 µg/dL are associated with various indices of reduced cognitive function and increased incidence of attention-related and problem behaviours in children (NTP 2012). The consistency of the effects observed at BLLs < 5 µg/dL is most evident from studies in children 3 to 18 years old (Braun *et al*. 2006; Chiodo *et al*. 2004; 2007; Cho *et al.* 2010; Froelich *et al.* 2009; Ha *et al*. 2009; Kim *et al*. 2010; Nicolescu *et al.* 2010; Nigg *et al*. 2008; 2010; Plusquellec *et al*. 2010; Wang *et al*. 2008). Two studies that employed the same NHANES dataset (1999–2002) reported similar results at a mean BLL of ≈ 2 µg/dL (Braun *et al*. 2006; Froelich *et al*. 2009), whereas Nigg *et al.* (2010) reported an association between BLL and ADHD at a mean BLL of 0.73 µg/dL (max = 2.2 µg/dL).

The adverse effects of early-life lead exposure on psychometric intelligence (IQ) tests are the most studied endpoint and have the greatest weight of evidence connecting lead exposure in children to adverse neurodevelopmental outcomes. Some jurisdictions (CalEPA 2009; HC 2019*a*) considered that an average 1-IQ point loss at the population level to be associated with significant public health implcations. Epidemiological evidence in a wide variety of ethnic and socioeconomic populations across a wide range of lead exposures generally indicates effects of concern at BLLs < 10 µg/dL (Bellinger *et al*. 1991; 1992; Cooney *et al*. 1989*a*; *b*; 1991; Dietrich *et al*. 1992; Ernhart *et al*. 1987; 1989; Schnaas *et al*. 2006). A number of epidemiological studies failed to identify a threshold for neurodevelopmental toxicity at BLLs as low as 1 to 2 µg/dL (Canfield *et al.* 2003*a*; Chiodo *et al.* 2004; 2007; Lanphear *et al.* 2000; 2005; Schnaas *et al.* 2006; Schwartz 1994; Tellez-Rojo *et al.* 2006), but not all studies report this pattern of results (Chandramouli *et al.* 2009; Min *et al.* 2009; Surkan *et al.* 2007). While the data on effects < 5 µg/dL provide strong evidence of effects at low BLLs, technological measurement limitations preclude the use of these levels as clear thresholds of effect. Evidence also indicates the persistence, until at least the late teenage years, of neurological effects associated with childhood BLLs, including effects on memory, learning (IQ), attention, visual construction[4], and fine-motor coordination (Fergusson *et al*. 1997; Ris *et al*. 2004). However, the literature lacks longitudinal studies in adults to confirm whether the effects are irreversible.

Occupational cohort studies have relatively consistently reported a number of central- and peripheral nervous system detrimental effects associated with BLLs > 20 µg/dL, with limited data available for studies at lower BLLs. Effects reported in occupational cohort studies include

[4] Assessed through the Block Design Subtest of the WISC-III and the Rey-Osterrieth Complex Figure Accuracy Score.

abnormal postural sway, abnormal visual-evoked potential and brainstem auditory-evoked potential, peripheral sensory nerve impairment, neuromotor impairment, and neurological symptoms (ATSDR 2007*a*). Aberrant electroencephalograph readings were significantly correlated ($p < 0.05$) with BLLs > 15 µg/dL in children, with non-significant effects at levels as low as 6 µg/dL (Otto *et al*. 1981; 1982). The auditory nerve may be a target for lead toxicity, based on reports of reduced hearing acuity in children (Robinson *et al*. 1985; Schwartz and Otto 1991).

The health effects reported in the human observational studies are largely supported by experimental studies in laboratory animals. While there is no equivalent to an IQ test for animals, adverse behavioural outcomes that reflect learning and memory have been demonstrated in multiple species, including non-human primates, at exposures resulting in BLLs of ≈ 10 µg/dL (Cory-Slechta and Thompson 1979; Gilbert and Rice 1987; Rice 1984; 1985; Rice and Gilbert 1985; Rice and Karpinski 1988). *In vivo* studies also indicate that the neurodevelopmental toxicity of lead persists after exposures cease (Cory-Slechta 1995; Rice 1984; 1985; Rice and Barone 2000). Furthermore, in rats, relatively high exposure causes adverse effects on both the retina and the optic nerve, whereas lower exposures primarily affect rod photoreceptors and bipolar cells (Fox and Boyes 2008). In rats, gestational exposures to low lead levels that produced peak BLLs of at least 12 µg/dL were associated with supernormal electroretinograms (ERGs), increased rod neurogenesis and decreased retinal dopamine (Fox and Boyes 2008). Higher exposures, producing peak BLLs of at least 46 µg/dL, induced further loss of retinal dopamine and rod cells, as well as ERG subnormality. These deficits are associated with an increased risk of spatio-temporal contrast sensitivity deficit and are consistent with observations in 7- to 10-year-old humans, where gestational lead exposure in an environmentally exposed cohort was associated with supernormal scotopic ERGs (Rothenberg *et al.* 2002*b*).

Varying evidence supports a wide range of mechanisms that could result in impaired cellular functioning and survival leading to impaired central nervous system function, including induced apoptosis. Studies investigating lead-induced apoptosis in rod cells suggest that lead depolarizes mitochondria, resulting in cytochrome *c* release and the activation of the caspase cascade (Fox and Boyes 2008; Fox *et al*. 1991; 1997; He *et al*. 2000; Srivastava *et al*. 1995), decreased cellular respiration (Li *et al.* 2007), disruption of neurotransmitter synthesis, storage, release, and reuptake (Lasley and Gilbert 2002), oxidative stress (Li *et al.* 2007), zinc and calcium mimicry (Pb^{2+} can replace these essential metals in metallo-enzymes), with associated disruption in homeostasis and protein function (Hanas *et al.* 1999; Kern and Audesirk 2000; Lidsky and Schneider 2003; Tomsig and Suszkiw 1995; Zawia *et al.* 1998), alteration of protein kinase C activity (Cremin and Smith 2002), impaired synaptic plasticity (Gilbert *et al.* 2005; Verina *et al*. 2007), disturbance of glial-cell functioning (Coria *et al.* 1984; Deng *et al.* 2001; Qian *et al*. 2005; 2007; Sierra and Tiffany-Castiglioni 1991), disruption of the normal function of the hypothalamic-pituitary-adrenal axis and related interaction with the mesocorticolimbic dopamine system (Cory-Slechta *et al*. 2004; Virgolini *et al*. 2005), and alterations to the epigenome (Basha *et al*. 2005; Wu *et al*. 2008). Lead has also been shown to pass readily through the blood-brain barrier via Ca^{2+}-ATPase pump uptake by brain capillary endothelial cells (Lidsky and Schneider 2003).

5.5.1.1 Effects on Children's Intelligence Quotient

Several studies (Emory *et al*. 2003; Schnaas *et al*. 2006; Wasserman *et al*. 2000*a*) suggest that prenatal maternal BLLs are correlated with IQ deficits in young children—effects that were significant when child BLLs were included as a potential confounder. In contrast, work by Lanphear *et al*. (2005) determined that umbilical cord BLLs were not correlated with IQ deficits when the children's BLLs were included as a potential confounder. The analysis of effects in children associated with maternal BLLs is complicated by lead exposures during childhood, which are independent of maternal exposure.

5.5.1.2 Neurodegenerative Effects

Observational and experimental evidence (Khalil *et al.* 2009; Stewart and Schwartz 2007; Weuve *et al*. 2009; Wright *et al*. 2003) supports an association between lead exposure and increased neurological decline. However, the weight of evidence is limited for BLLs < 10 µg/dL (HC 2013*a*). An association between lead exposure and cognitive decline has been reported at mean BLLs as low as 5 µg/dL (Wright *et al*. 2003); however, findings from other cohort studies examining both men and women with similar exposures did not observe this effect (Krieg *et al*. 2005; Muldoon *et al*. 1996; Nordberg *et al*. 2000).

Neurodegenerative effects are more consistently associated with PbBO concentrations than BLLs (Shih *et al*. 2006). Tibia lead, but not blood lead, has been associated with cognitive decline in a cohort of older women with BLLs of 2.9 µg/dL (Weuve *et al*. 2009) and with lower current cognitive performance and cognitive decline in a longitudinal occupational cohort study (Khalil *et al*. 2009).

Few experimental data demonstrate the effects of lead-induced neurodegeneration on animal behaviour. Studies in both rats and non-human primates at BLLs of ≈ 20 µg/dL demonstrate that older animals are more susceptible to the adverse neurobehavioural effects of lead than younger animals (Cory-Slechta and Pokora 1991; Rice 1990; 1992*a*; *b*).

5.5.2 Cardiovascular Effects

The scientific literature contains multiple lines of evidence, including human epidemiological studies, *in vivo* animal assays and *in vitro* experiments, that chronic lead exposure can result in adverse cardiovascular effects by various mechanisms. Increased rates of cardiovascular morbidity and mortality have been observed in humans at exposure levels resulting in BLLs < 10 µg/dL (Lustberg and Silbergeld 2002; Menke *et al*. 2006; Navas-Acien *et al*. 2004; 2007; 2008; Schober *et al*. 2006). The most-studied endpoints, for which there is the greatest weight of evidence of a causal relationship, are lead-induced increases in blood pressure, particularly systolic blood pressure (SBP), and the associated risk of hypertension.

While environmental cohort study data show an increase in SBP or the risk of hypertension in adults with BLLs as low as 3–5 µg/dL (Martin *et al.* 2006; Vupputuri *et al.* 2003), others have disputed the presence of a relationship between low-level lead exposure and hypertension (Nawrot

et al. 2002; Schwartz 1995; Staessen *et al*. 1994). Analyses of the BLL-response relationship curve show evidence of an attenuation of the slope at the low end of current BLLs for SBP and myocardial infarction mortality (Martin *et al*. 2006; Menke *et al*. 2006; Schwartz and Stewart 2000), but not stroke mortality (Menke *et al*. 2006). Meta-analyses of the epidemiological findings identified a persistent trend that supports a relatively mild, but statistically significant, association between BLLs and SBP in adults (ATSDR 2007*a*; EFSA 2013; ILZRO and EBRC 2008). Several studies in rats show significant increases in blood pressure at BLLs < 10 µg/dL after oral lead exposure (Ding *et al.* 1998; 2001; Khalil-Manesh *et al.* 1994; Vaziri *et al.* 1999*a*; *b*).

The modest effects and inconsistency of observed results can be attributed, in part, to lead exposure and blood pressure measurement error. Other risk factors of cardiovascular disease, such as age, diet, gender, and ethnicity, may also contribute to the inconsistency in the reported results. In contrast, PbBO has been more consistently associated with increased blood pressure or risk of hypertension in the aged (Cheng *et al*. 2001). Epidemiological evidence is insufficient to support an association between lead exposure and increased blood pressure in children. No published studies on the relationship between PbBO and blood pressure in children were identified.

The collective evidence from epidemiological and *in vivo* and *in vitro* experimental studies clearly indicates several mechanisms by which chronic lead exposure could cause elevated blood pressure and related cardiovascular disease. The mode of action for which there is the most evidence is lead-induced oxidative stress, leading to the inactivation of the vasodilator nitric oxide as well as related signalling pathways and functional responses. These effects can increase blood pressure and contribute to hypertension (Courtois *et al.* 2003; Farmand *et al.* 2005; Khalil-Manesh *et al.* 1993*a*; Ni *et al.* 2004; Vaziri *et al.* 2001). Lead is also known to activate nuclear transcription factors in the brain, which can lead to inflammation and contribute to several other effects, including: hypertension (Carmignani *et al.* 2000; Ramesh *et al.* 2001; Rodriguez-Iturbe *et al.* 2005); stimulation (or inhibition of suppression) of the adrenergic system, leading to increased blood pressure (Carmignani *et al.* 2000; Chang *et al.* 1997; Tsao *et al.* 2000); increases in protein kinase C activity, which regulates vascular tone and blood flow (Lidsky and Schneider 2003; Valencia *et al.* 2001; Watts *et al.* 1995); direct or indirect (through stimulation of the sympathetic nervous system) induction of the renin-angiotensin-aldosterone and kininergic systems (Carmignani *et al.* 1999); induction of paracrine systems involved in blood pressure regulation (Bagchi and Preuss 2005; Boscolo and Carmignani 1988; Carmignani *et al.* 1999); disruption of vasodilator and vasoconstrictor prostaglandin balance (Cardenas *et al*. 1993; Gonick *et al*. 1997; Hotter *et al*. 1995); increase in plasma endothelin levels, associated with lead-induced hypertension (Gonick *et al.* 1997; Khalil-Manesh *et al.* 1993*b*; 1994); inhibition of endothelial cell proliferation and impairment of angiogenesis and endothelial cell repair (Fujiwara *et al.* 1998; Kaji *et al.* 1995; Ueda *et al.* 1997); and inhibition of Na-K-ATPase activity, associated with hypertension and increases in blood pressure (Glenn *et al*. 2001).

5.5.3 Renal Effects

Elevated BLLs have long been associated with adverse renal outcomes. Historically, chronic nephropathy was not detected in adults and children with BLLs < 40 µg/dL (Campbell *et al*. 1977; Lilis *et al*. 1977). In a review of the epidemiological literature, Ekong *et al*. (2006) concluded that

lead contributes to nephrotoxicity in individuals with BLLs < 5 µg/dL, particularly in susceptible populations, such as those with hypertension, diabetes and/or chronic kidney disease. An increased risk of renal dysfunction and rate of functional decline, reduced glomerular filtration rate, and reduced creatinine clearance were associated with BLLs as low as 2 µg/dL, after adjusting for confounders, in several adult cohorts (Akesson *et al.* 2005; Muntner *et al.* 2003; Tsaih *et al.* 2004; Yu *et al.* 2004). Staessen *et al*. (1992), Payton *et al*. (1994) and Kim *et al*. (1996) noted positive associations between BLLs and creatinine levels in exposed subjects. When the data were stratified by disease status, a stronger inverse longitudinal relationship between BLLs or PbBO levels and creatinine clearance was reported in hypertensives and diabetics, which suggests possible interactions between lead exposure, glomerular function, and diabetes or hypertension (ATSDR 2007*a*; Muntner *et al.* 2003; Tsaih *et al.* 2004).

Few data exist on environmentally relevant (non-occupational) lead exposures and renal function in other mammals. In rats, BLLs of 26 µg/dL were associated with increased creatinine clearance and accelerated renal microvascular and tubulointerstitial injury (Roncal *et al.* 2007). Similar BLLs (29 µg/dL) in rats were associated with transient hyperfiltration, which is consistent with evidence from relatively high-exposure occupational cohort studies showing mild tubular atrophy and interstitial fibrosis (Khalil-Manesh *et al.* 1993*b*).

Lead has been reported to cause proximal tubular injury, with a characteristic pathology of proximal tubule nuclear inclusion bodies progressing to tubulointerstitial disease and fibrosis (Khalil-Manesh 1993*b*). Lead accumulation in the proximal tubule leads to hyperuricemia and gout, presumably by inhibiting uric acid secretion and diminishing renal clearance, tubular reabsorption and glomerular filtration rate (Gonick 2008). In humans, severe nephritic deficits in function and pathological changes are observed at BLLs > 50 µg/dL, enzymuria and proteinuria become evident at BLLs > 30 µg/dL, and reduced glomerular filtration (measured as a decrease in creatinine clearance or increase in serum creatinine) at BLLs < 20 µg/dL (ATSDR 2007*a*).

5.5.4 Reproductive Effects and Teratogenicity

5.5.4.1 Male Reproductive Effects

Relatively high developmental (*in utero*) lead exposures can delay the onset of puberty and alter reproductive function in males (HC 2013*a*). Reported male reproductive effects include decreased sperm count, morphological aberrations and an increased risk of infertility (Alexander *et al*. 1996; Bonde *et al*. 2002; Sallmen *et al*. 2000). While the majority of the literature reports effects at occupational exposure levels (10–50 µg/dL) (Sallmen *et al.* 2000), one study reported decreased growth and differences in puberty onset at BLLs as low as 3 µg/dL (Hauser *et al.* 2008).

In vivo data from test animals has demonstrated that BLLs > 30 µg/dL are associated with delayed sexual maturation, reduced semen quality, morphological and histological changes in sex organs and impaired fertilization (ATSDR 2007*a*). In monkeys, BLLs as low as 20 µg/dL suppressed gonadotropin-releasing hormone–induced secretion of luteinising hormone (LH), decreased testosterone:LH ratio, and altered Sertoli cell function (Foster *et al.* 1993). Structural effects in Sertoli cells and spermatids, as well as reduced numbers of spermatozoa, were observed in rats at

BLLs of ≈ 7 µg/dL (Barratt *et al.* 1989; Murthy *et al.* 1995). BLLs of ≈ 14 µg/dL in rats were associated with structural damage to seminiferous tubules and a reduced number of prospermatogonia (Corpas *et al.* 1995).

5.5.4.2 Female Reproductive and Teratogenic Effects

Key female reproductive effects associated with lead exposure include delayed sexual maturation, risk of spontaneous abortion, and low birth weight and/or pre-term birth (HC 2013*a*). Reduced fertility and an increase in the frequency of miscarriage and stillbirth are considered common clinical reproductive manifestations among women who have been occupationally exposed to high lead levels (BLLs in the range of 30–40 µg/dL) (ATSDR 2007*a*; Flora and Tandon 1987; Hertz-Picciotto 2000). Some studies have shown an association between increased BLLs and spontaneous abortion (Borja-Aburto *et al*. 1999; Lamadrid-Figueroa *et al*. 2007). However, in another study, no significant difference was reported in mean BLLs between spontaneous abortion cases and ongoing pregnancies (Vigeh *et al*. 2010). Overall, there is limited evidence for the increased risk of spontaneous abortions at BLLs < 30 µg/dL (CDC 2010; HC 2013*a*).

Findings from environmental cohort studies investigating associations between maternal blood and/or PbBO levels and birth weight and pre-term birth are inconsistent, and methodological issues limit the strength of the conclusions we can draw from them (CDC 2010; HC 2013*a*). However, several studies show an inverse relation between maternal PbBO and birth weight (Gonzalez-Cossio *et al*. 1997), birth length (Hernández-Avila *et al*. 2002), and the length of gestation (Cantonwine *et al*. 2010).

Epidemiological evidence supports an association between BLL > 2 µg/dL and delayed puberty in adolescent girls (Flannery *et al*. 2020; Naicker *et al*. 2010; Selevan *et al.* 2003). This finding was replicated in two of the three additional studies that examined this effect (Denham *et al.* 2005; Wolff *et al.* 2008; Wu *et al.* 2003), and supported by *in vivo* animal studies in mice (Iavicoli *et al.* 2004; 2006), monkeys (at higher BLLs of 25–30 µg/dL) (Foster *et al.* 1996), and rats (Dearth *et al*. 2002). However, studies showing no effects limit the weight of evidence for these effects (Flannery *et al*. 2020).

5.6 Overall Toxicological Evaluation

The information in the following sections draws heavily from the HC (2013*a*)'s *Final Human Health State of the Science Report on Lead*, the toxicological review compiled by Healey *et al* (2010), and Wilson and Richardson (2013). Reported effects associated with BLLs below the Canadian blood lead intervention level of 10 µg/dL (Committee on Environmental and Occupational Health 1994) include neurodevelopmental, neurodegenerative, cardiovascular, renal, and reproductive effects (Canfield *et al*. 2003*a*; *b*; Chandramouli *et al*. 2009; Chiodo *et al*. 2004; 2007; Després *et al*. 2005; Fraser *et al*. 2006; Lanphear *et al*. 2000; Miranda *et al*. 2007; Osman *et al*. 1999). Several studies report a dose-response relationship that extends down to the lowest studied BLL concentrations (1–2 µg/dL) (Canfield *et al*. 2003*a*; Chiodo *et al*. 2004; Jedryschowski *et al*. 2009; Jusko *et al*. 2008; Lanphear *et al*. 2005; Miranda *et al*. 2007; Tellez-Rojo *et al*. 2006). Dose-response modelling conducted with available data does not currently demonstrate a

population threshold for developmental neurotoxicity. The weight of evidence for effects in this range of exposure is strongest for neurodevelopmental effects in children, most commonly assessed as the reduction of intelligence quotient (IQ) score with increasing BLLs (ATSDR 2007*a*; California Office of Environmental Health Hazard Assessment [CalOEHHA] 2007; EFSA 2013; HC 2013*a*, 2019*a*; WHO/JECFA 2011). The epidemiological evidence demonstrating a correlation between BLLs and neurodevelopmental toxicity strongly suggests an adverse association between early-life chronic lead exposure and decrements in school-aged children's IQ. However, note that for many of the studies, the limit of quantification was between 1 and 3 μg/dL, which makes it difficult to interpret the lower end of the dose-response curve. Overall, the weight of evidence from observational and *in vivo* experimental animal studies supports the following conclusions:

- Neurodevelopmental toxicity associated with the lowest reported levels of lead exposure, both in observational studies and in animal experiments.
- In humans, the developmental neurotoxic effects of lead can persist until the late teenage years.
- In laboratory animals, the developmental neurotoxic effects of lead can persist after exposures have ceased and blood and brain lead concentrations have returned to normal or control levels.
- The preponderance of data from observational studies does not demonstrate a population threshold for neurodevelopmental toxicity over the lower ranges of current environmental lead exposures.
- Lead can interact with multiple cell types in the central nervous system, and potential modes of action supported by experimental evidence have been developed to explain lead's observed neurodevelopmental toxicity. These modes of action are considered relevant to humans.

In adults, the most-studied endpoint and the one with the greatest weight of evidence of a causal relationship is cardiovascular toxicity; specifically, lead-induced increases in blood pressure (ATSDR 2007*a*; EFSA 2013; HC 2013*a*; WHO/JECFA 2011). The epidemiological literature supports a relatively small but statistically significant association between BLLs and increases in blood pressure, particularly SBP in adults (Glenn *et al*. 2003; 2006; Lee *et al*. 2001; Nash *et al*. 2003; Nawrot *et al*. 2002; Schwartz 1995; Staessen *et al*. 1994; Vupputuri *et al*. 2003). Quantitatively, each doubling of BLL is associated with an approximate 1 millimeter of mercury (mmHg) increase in SBP (Nawrot *et al*. 2002; Schwartz and Stewart 2000; Staessen *et al*. 1994). Overall, there is suggestive, but not entirely consistent, epidemiological evidence of an association between environmental lead exposure and SBP or risk of hypertension among subjects with average BLL < 10 μg/dL.

5.7 Toxicological Limits

As part of the overall toxicological evaluation, CCME assessed available exposure limits for human health effects and toxicological data to determine one or more critical studies upon which to develop an exposure limit to use in deriving a $PSoQG_{HH}$.

CCME has developed distinct methods for deriving SoQGs applicable to substances with threshold and non-threshold modes of toxic action (CCME 2006). While evidence suggests that lead exposure may be associated with cancer development in humans, an evaluation of available toxicological data and dose-response information indicates non-cancer endpoints are more sensitive. Although a non-cancer endpoint was identified as the most sensitive endpoint, a threshold for effects has not been identified for lead. Therefore, a tolerable daily intake was not identified and government agencies have concluded that lead should be treated as a non-threshold substance (Advisory Committee on Childhood Lead Poisoning Prevention 2012; CalEPA 1997; 2009; HC 2013*a*; *b*; US EPA 2006*a*; WHO/JECFA 2011) based on the available scientific literature.

5.7.1 Historical Lead Exposure Limits

HWC (1987) recommended a maximum acceptable BLL of 25 µg/dL (1.2 µmol/L) for the protection of all Canadians, based on a 40 µg/dL threshold for adverse clinical effects in males and considering that women, children and fetuses were more sensitive to the toxic effects of lead.

In 1994, this guidance level was updated based on a Federal-Provincial Advisory Committee on Environmental and Occupational Health report that recommended a blood intervention level of 10 µg/dL based on evidence of health effects in the range of 10–15 µg/dL (HC 1994). This recommendation was adopted, and Canada's blood lead guidance level was updated to match the World Health Organization's (WHO/JECFA 1986) provisional tolerable weekly intake (pTWI) of 25 µg/kg body weight per week (equivalent to a provisional tolerable daily intake [pTDI] of 3.57 µg/kg bw/d), which were designated "provisional" due to the absence of a clear threshold of effects for lead. This position is consistent with statements by the ATSDR (2007*a*), CalEPA (1997), CalOEHHA (2001), OMEE (1994) and WHO/JECFA (2000).

HC (2013*a*)'s research has conclusively demonstrated that BLLs < 10 µg/dL can produce adverse health outcomes in both children and adults. Therefore, HC no longer supports the use of the pTDI of 3.57 µg/kg bw/d. In 2010 the WHO/JECFA concluded that the pTWI for lead could no longer be considered protective of health and withdrew it (2011).

5.7.2 Critical Toxicological Endpoints

The selected endpoints (IQ decrements and SBP) have been extensively studied and data are available to quantitatively assess the adverse effects and their relationship to BLLs < 10 µg/dL. For the purpose of the development of $PSoQGs_{HH}$, RSDs were derived based on a novel approach, using BLL, and a non-threshold approach for a non-carcinogenic substance. For these reasons and because no lower threshold of effects has been identified for the critical effect, the RSDs are considered provisional and therefore, the SoQGs developed using these values are considered provisional ($PSoQGs_{HH}$). The approach used for lead also considers potential risks to infants and adolescents, for whom there are currently insufficient data to derive limits.

5.7.2.1 Neurodevelopmental Toxicity in Children

Intellectual function, as measured by full-scale Wechsler IQ, was selected as the relevant toxicological indicator because it is considered a sensitive marker for neurodevelopmental effects and is the most scientifically supported toxicological endpoint for lead in children (ATSDR 2007*a*; CalOEHHA 2007; EFSA 2013; WHO/JECFA 2011).

The majority of cross-sectional and longitudinal studies using IQ measurements or the McCarthy Scales of Children's Ability General Cognitive Index report an inverse relationship between BLLs and measured cognitive function (e.g., Canfield *et al*. 2003*a*; *b*; Fulton *et al*. 1987; Lanphear *et al*. 2000; 2005).

In the most comprehensive analysis of neurodevelopmental toxicity, Lanphear *et al*. (2005) pooled and analysed data from seven longitudinal prospective studies of 1,333 children from around the world. This large sample size provided the statistical power needed to characterize the relationship between BLLs and IQ scores. The studies were initiated before 1995 and followed subjects from birth or infancy until 5 to 10 years of age in Boston, Massachusetts (Bellinger *et al*. 1992); Cincinnati, Ohio (Dietrich *et al*. 1993); Cleveland, Ohio (Ernhart *et al*. 1989); Rochester, New York (Canfield *et al.* 2003*a*); Mexico City, Mexico (Schnaas *et al*. 2000); Port Pirie, Australia (Baghurst *et al*. 1992); and Kosovo (Wasserman *et al*. 1997). Venous or capillary blood samples were analysed for four blood lead indices: 1) concurrent, 2) maximum, 3) lifetime average, and 4) early childhood (defined as 6 months to 2 years) mean blood lead. The pooled median concurrent (i.e., BLLs measured closest to the time of cognitive testing), maximum, lifetime-average, and early-childhood BLLs were 9.7, 18.0, 12.4, and 12.7 µg/dL, respectively. Of the 244 subjects, 18% had maximum BLLs < 10 µg/dL and 8% (n = 103) had a maximum BLLs < 7.5 µg/dL. The covariates included maternal IQ, education, marital status, prenatal alcohol, and tobacco use, Home Observation and Measurement of the Environment inventory score, subject sex, birth order, and birth weight. The influence of ethnicity was investigated for the subset of US data. Potentially important covariates that were not included were socioeconomic status, nutritional status, and paternal IQ.

Lanphear *et al*. (2005) found an inverse relationship between concurrent BLL and IQ score, and concluded that intellectual effects occur in children with maximal BLLs of < 7.5 µg/dL.

Based on the data set used by Lanphear *et al*. (2005), agencies such as the CalEPA, EFSA and WHO/JECFA conducted dose-response analyses to characterize the neurodevelopmental risk of lead exposure in children. CalEPA estimated that a 1 µg/dL increment in BLL was associated with a 1-IQ point deficit (lower 97.5th percentile estimate) (CalEPA 2009; CalOEHHA 2007). EFSA (2013) estimated that a 1.2 µg/dL increment in BLL was associated with a 1-IQ point decrement (lower 95th percentile estimate). WHO/JECFA (2011) concluded that a 2 µg/dL increment in BLL would be associated with a 1-IQ point decrement (central estimate).

Using a 1.2 µg/dL change in BLL, EFSA (2013) concluded that a dose of 0.5 µg/kg bw/d would be associated with a population-level 1-IQ point decrement in children. Applying a piecewise linear model to estimate a benchmark dose, EFSA (2013) determined that a 1.8 µg/dL increment in BLL (benchmark dose for a 1% additional risk [BMD_{01}]) was associated with a 1-IQ point

decrement, and the 95th percentile lower confidence limit of the benchmark dose lower bound ($BMDL_{01}$) was 1.2 µg/dL. Applying a toxicokinetic relationship, whereby an intake rate of 1 µg/kg bw/d of lead results in a 2.4 µg/dL BLL, EFSA (2013) estimated that a $BMDL_{01}$ of 0.5 µg/kg bw/d represents a 95th percentile lower confidence limit of the estimated BMD associated with a 1-IQ point decrement.

On the other hand, WHO/JECFA (2011) concluded that a dose of 0.6 µg/kg bw/d resulted in a population-level 1-IQ point decrement in children. Although WHO/JECFA (2011) considered a BLL of 2 µg/dL was associated with a 1-IQ point decrement, (i.e., less conservative than EFSA [2013]), they applied a more stringent toxicokinetic approach to estimate intake rates associated with a 1-IQ point decrement. Therefore, the intake rate of 0.6 µg/kg bw/d associated with a 1-IQ point decrement, based on the WHO/JECFA (2011) approach, is similar to the EFSA (2013) value for a 1-IQ point decrement.

Overall, CCME considers the EFSA (2013) analysis to be the most rigorous. While quite similar to the WHO/JECFA (2011) analysis, it had the additional benefit of a protection goal of a 95th percentile lower confidence limit rather than a central estimate. EFSA (2013) is also more consistent with the HC (2013*a*; 2013*b*) conclusion that an incremental increase in BLL of 1 µg/dL is associated with a decrement of approximately 1-IQ point. Therefore, based on the EFSA (2013) analysis, CCME considers an intake rate of 0.5 µg/kg bw/d to be an RSD associated with a 1-IQ point decrement in infants, toddlers, and children.

WHO/JECFA (2011) determined the relationship between BLLs and dietary exposure, estimating mean dietary exposures ranging from 0.03 to 9 µg/kg bw/d for children aged about 1–4 years. WHO/JECFA (2011) considered the health impact at the lower end of the range to be "negligible" because it is below the exposure level of 0.3 µg/kg bw/d calculated to be associated with a population-level decrease of 0.5 IQ point.

Given that no threshold has been observed for developmental neurotoxicity over the lower ranges of environmental exposure, it is prudent to reduce exposure to lead (and associated risks) to the greatest extent practicable (HC 2013*b*).

Consequently, for the purpose of derivation of the $PSoQGs_{HH,}$ CCME is presenting two options, one using the EFSA (2013) intake of 0.5 µg/kg bw/d as an RSD targeting a 1-IQ point decrement and one using the RSD with a 50% adjustment factor to target a 0.5-IQ point decrement. An IQ decrement of 0.5 was considered by WHO/JECFA (2011) to be the upper end of the negligible range. CCME considers the addition of an adjustment factor appropriate, even though the correlation between critical effects and BLL (hence target IQ decrements) is acknowledged to be non-linear.

5.7.2.2 Cardiovascular Effects in Adults

The cardiovascular biomarker endpoint for which there is the greatest weight of evidence of a causal relationship is increased blood pressure, particularly SBP (ATSDR 2007*a*; EFSA 2013; HC 2013*a*; *b*; WHO/JECFA 2011). As a whole, meta-analyses of the epidemiological findings have identified a persistent trend that supports a relatively mild, but statistically significant, association

between BLLs and SBP in adults (ATSDR 2007*a*; EFSA 2013; HC 2013*a*; *b*; ILZRO and EBRC 2008; WHO/JECFA 2011). Furthermore, ATSDR (2007*a*) has indicated that long-term exposure to lead can elevate blood pressure in rats. PbBO has more consistently been associated with increased blood pressure or risk of hypertension (Navas-Acien *et al.* 2008).

SBP is used to assess the effects of lead exposure in adults because the effect is stronger, it is more frequently associated with elevated lead body burdens than diastolic blood pressure, and it may be a more important risk factor for cardiovascular mortality. A 1.1 mmHg increase in mean population SBP was defined as the adverse-effect response level for determining the adult TRV (1% of the population mean [111.1 mmHg], measured from 2007–2009 in Canadian women aged 20–79 years [Wilkins *et al.* 2010]).

Work by Vupputuri *et al.* (2003) and Den Hond *et al.* (2002) suggests that females and African Americans may be more sensitive to SBP effects, although it should be noted that Caucasians (18 years and older) generally had lower BLLs. In the most recent and comprehensive published review of SBP effects, WHO/JECFA (2011) averaged the linear slope estimates relating SBP increases as a function of BLLs from four studies (Glenn *et al*. 2003; 2006; Nash *et al*. 2003; Vupputuri *et al*. 2003) to derive a median slope estimate of 0.28 mmHg per 1 µg/dL. BLLs were then converted to dose levels associated with a 1 mmHg increase in SBP. CCME estimated a dose level of 80 µg/d or 1.3 µg/kg bw/d (5^{th}-95^{th} percentile = 0.6–28 µg/kg bw/d), and selected this RSD, protective of a 1 mmHg SBP increase, for the purposes of $PSoQG_{HH}$ development. This TRV is considered provisional and protective on an individual risk basis.

5.7.2.3 Women of Childbearing Age and Sensitive Subpopulations

Both child and adult sensitive subpopulations have been identified as sensitive subpopulations for the effects of lead exposure. It is important that the selected TRVs address the most sensitive of these subpopulations. Sensitive adult populations for cardiovascular effects may include African Americans (Den Hond *et al.* 2002; Vupputuri *et al.* 2003), pregnant women (Rothenberg *et al.* 2002*a*), postmenopausal women (Nash *et al.* 2003), and hypertensive or diabetic elderly people (Tsaih *et al.* 2004).

Bradman *et al.* (2001) determined that lower socioeconomic status is associated with greater BLLs in children, which could be due to lower iron or calcium intake. Muldoon *et al.* (1996) and Schneider *et al.* (2001) suggested that socioeconomic status and education level might limit the extent of lead-related neurocognitive deficits. Infants and children are a susceptible subpopulation because they have greater gastrointestinal absorption and less effective renal excretion than adults do. The selection of infants and children as susceptible subpopulations and neurodevelopmental effects as the critical health effect is considered protective for other adverse effects of lead across the entire population (HC 2013*a*; *b*).

The developing fetus is in a sensitive life stage. Wilson and Richardson (2013) noted that the 1 mmHg increase in SBP risk level developed by WHO/JECFA (2011) for the protection of adults should also be protective of fetuses and women of childbearing age. However, the US EPA (2003) concluded that adults absorb lead at a lower rate than children do and neither EFSA (2013) nor

WHO/JECFA (2011) provide specific conclusions as to whether the proposed adult intake rates would be protective of the developing fetus. Therefore, CCME adopted the EFSA (2013) 0.5 µg/kg bw/d intake rate to protect all age groups, including women of childbearing age for all land uses. For the purpose of derivation of the $PSoQGs_{HH}$ CCME is presenting two options, one using the RSD of 0.5 µg/kg bw/d to target a 1-IQ point decrement and one using a 50% adjustment factor applied to the equation in order to target a 0.5-IQ point decrement, for the protection of adults, including women of childbearing age.

5.7.3 Application of Uncertainty Factors

Based on the analysis conducted by EFSA (2013), no uncertainty factors were applied to the TRVs for the protection of potentially sensitive children or adult subpopulations; instead, CCME used a non-threshold approach.

The EFSA (2013) $BMDL_{01}$ of 0.5 µg/kg bw/d represents a 95th percentile lower confidence limit of the BMD associated with a 1-IQ point decrement, which should be interpreted as the intake rate associated with a drop of 1 IQ point in a population of children. In addition to this interpretation, the key authors of EFSA (2013) published additional information after applying a hybrid model that was not central to the EFSA (2013) analysis. Budtz-Jørgensen *et al*. (2013) indicated that the 1.2 µg/dL BMDL (or 0.5 µg/kg bw/d) is also associated with a 1% increased risk of having an IQ of less than 75 points.

Based on CalEPA's (2009) assessment that a 1-IQ point decrement was likely insignificant from an individual perspective but could be biologically and statistically significant from a population perspective, CCME considered extrapolation to a slightly lower risk level to also be appropriate to target an IQ decrement less than 1. Hence, two guidelines were developed, one using the RSD associated with a 1-IQ point decrement and one using the RSD associated with a 0.5-IQ point decrement, to provide options for contaminated site management.

5.7.4 Level of Confidence and Uncertainties

The level of confidence is high in the scientific literature reporting the association between lead exposure and neurodevelopmental toxicity in humans. Numerous human observational studies assessing multiple organs or systems are available. Furthermore, the critical health effects identified are based on well-established endpoints and are supported by mechanistic data as well as studies conducted in laboratory animals. Despite some uncertainties, the overall findings from the literature are relatively strong and clear, particularly regarding the most critical health effect identified in children.

The following uncertainties were identified with respect to the available data for lead. The selected RSDs for lead in children are based on the EFSA (2013) potency estimate, in which a dose of 0.5 µg/kg bw/d is associated with a 1-IQ point decrement (roughly equivalent to a BLL of 1.2 µg/dL). This estimate largely relies on the Lanphear *et al*. (2005) pooled analysis, and uncertainties in that analysis have been discussed in Wilson and Richardson (2013). Primary uncertainties relate to the quantification of effects at the low dose range (BLL < 2 µg/dL) due to

the small number of children in this exposure range in the Lanphear *et al*. (2005) study, and the lack of concurrence with other studies (such as Surkan *et al.* 2007) in which effects on IQ at BLL < 4 µg/dL were not reported.

As mentioned in Section 5.3.2, 96 to 99% of lead found in whole blood is complexed with proteins within red blood cells (Bergdahl *et al*. 1997; 1998; 1999; Hernandez-Avila *et al*. 1998; Manton *et al*. 2001; Schütz *et al*. 1996; Smith *et al*. 2002). While lead in whole blood is currently the most widely used index of lead exposure, several limitations must be addressed when interpreting the results of a BLL-health effect analysis (Skerfving *et al*. 1993). As a result, BLLs may not represent the most appropriate index of either exposure or effect. One noted limitation is the non-linear relationship between BLLs and lead exposure and gastrointestinal uptake; the increment in BLL per unit of lead intake decreases with increasing BLL, both in children (Lacey *et al*.1985; Rye *et al*. 1983; Sherlock and Quinn, 1986) and in adults (Kehoe 1987; Laxen *et al*. 1987; Pocock *et al*. 1983; Sherlock *et al.* 1982; 1984). The most probable explanation for this nonlinearity is the saturation of the red blood cells and the influence of PbBO (Skerfving *et al*. 1993; US EPA 2006*a*). The contribution of PbBO to BLLs can range from 40 to 70% of lead in whole blood. However, variations in the duration and intensity of lead exposure, as well as variations in physiological parameters such as sex, gender, diseases state, nutritional state, pregnancy, menopause and andropause, genetics, and ethnicity, can all influence lead uptake and resorption from bones, and the severity of these effects is not always accounted for in human studies (Gulson *et al*. 1995; Manton 1985; Smith *et al.* 1996; US EPA 2006*a*). While longitudinal measurements of BLLs may provide a reliable measure of an individual's exposure history (and will more closely parallel body burden) compared to a single BLL measurement, the degree to which this reliability applies depends on the sampling frequency with respect to the temporal pattern of exposure (US EPA 2006*a*). The use of BLLs as the biomarker of exposure is well correlated to health effects, but does not represent whole-body burden (e.g., lead sequestered into bone).

Furthermore, improper collection, handling, storage, and/or processing of blood samples can contribute to increased variability in serum lead measurements (Manton *et al*. 2000). Smith *et al.* (1998) demonstrated that the use of standard Vacutainer blood collection tubes artificially increased the measured serum lead value by four to six times over the values reported when ultraclean serum collection methods were employed. However, note that whole-blood concentration results were used in this analysis. Additionally, as with any biological test, whole-blood lead measurements have inherent uncertainties related to analytical limitations. When the seven studies analysed by Lanphear *et al*. (2005) were initiated, there were no globally accepted analytical protocols or standards, and the accepted US standard allowed for a ± 4 µg/dL variance in BLL measurements (CDC 2007). Additionally, note that the lowest data point in the Lanphear *et al*. (2005) study was 2.4 µg/dL. As such, there is uncertainty regarding the extrapolation of the dose-response curve to levels currently found in Canadians. The absence of lower data points in the Lanphear *et al*. (2005) study is related to its use of older studies conducted in times of higher ambient lead exposure.

Adverse effects observed in epidemiological studies may also be related, in part, to co-exposures to other chemicals. The TRVs for children and adults were developed from epidemiology data where co-exposure to other substances or stressors may have led to a chemical interaction (e.g., additivity, antagonism, synergism, transformation, potentiation), altering the shape and strength of

the dose-response relationship. However, while the effects observed were reported in numerous studies and substantiated by *in vivo* and *in vitro* toxicological data, the effects of chemical interactions on the toxicity of mixtures are not well understood, and in most cases guidelines are developed for single chemicals, which are treated as if they occur in isolation in the environment.

In addition, although overall IQ measures have been independently shown to be affected by lead exposure in children, IQ estimates computed from the Wechsler scales or other measures have not, in general, been demonstrated to be particularly sensitive to neurotoxic exposure (WHO 2001). While estimates of overall intellectual functioning are deemed necessary to adequately interpret deficits on particular tests in the context of a patient's general level of intellectual functioning, several subtests (e.g., Digit Symbol, Similarities and Digit Span) appear to be sensitive to neurotoxic exposures (WHO 2001).

5.7.5 Summary of the Toxicological Limits for Protection of Young Children

Since there is currently no identified threshold for the health effects of lead, a tolerable daily intake value (TDI) cannot be derived. When assessing risks posed by exposure to non-threshold substances, typically genotoxic carcinogens, it is necessary to specify a level of effect that is considered acceptable, tolerable, or essentially negligible. RSDs reflect regulatory policy that defines acceptable doses for incremental lifetime cancer risk levels less than or equal to 1×10^{-5} or 1×10^{-6}. Wilson and Richardson (2013) introduced the concept of developing $SoQG_{HH}$ for lead using the Protocol's non-threshold approach using RSDs derived based on non-cancer risk levels (EFSA 2013; WHO/JECFA 2011) for lead effects. This proposal is a non-conventional approach for a non-threshold substance for which the key endpoint is neurotoxicity in children, rather than carcinogenicity. In that context, Wilson and Richardson identified a 1-IQ point decrement as the "essentially negligible effect level" in their 2013 publication. Some jurisdictions (CalEPA 2009; HC 2019*a*) considered a 1-IQ point to be significant on a population level. CalEPA adopted a 1-IQ point decrement (predicted to be associated with a BLL of 1 µg/dL) to derive revised soil lead screening levels (CalEPA 2009).

WHO/JECFA (2011) determined the relationship between BLLs and dietary exposure, estimating the range of mean dietary exposures between 0.03 to 9 µg/kg bw/d for children aged about 1–4 years. WHO/JECFA (2011) considered the health impact at the lower end of the range to be "negligible" because it is below the exposure level of 0.3 µg/kg bw/d calculated to be associated with a population decrease of 0.5-IQ point. Therefore, in order to reflect the lower range of WHO/JECFA (2011), CCME provides $PSoQGs_{HH}$ that are also protective of a 0.5-IQ decrement.

Consequently, for the purpose of derivation of the $PSoQG_{HH}$, CCME is presenting two options, one for a 1-IQ point decrement and one using a 50% adjustment factor applied to the equation in order to target a 0.5-IQ point decrement for the protection of children.

5.7.6 Summary of the Toxicological Limits for Protection of Adults

With respect to adults, WHO/JECFA (2011) identified the critical endpoint as cardiovascular health (increases in SBP), and identified a population increase of 1 mmHg in SBP as a minimum

risk level. WHO/JECFA (2011) concluded that a dose rate of 1.3 µg/kg bw/d is associated with a 1 mmHg in SBP, while EFSA (2013) estimated that a dose rate of 1.5 µg/kg bw/d is associated with an increase of 1.2 mmHg in SBP. These dose estimates are very similar. The WHO/JECFA (2011) dose-rate estimate was selected as the toxicological limit (RSD) for SBP protection in adults. EFSA (2013) developed a separate potency estimate (RSD) of 0.63 µg/kg bw/d for the protection of renal effects.

Although an RSD for adults is available, this endpoint may not be protective the developing fetus. Therefore, of the EFSA (2013)-based RSD for the protection of developmental effects was adopted for women of childbearing age to protect developing fetuses. The same 50% adjustment factor that was applied for the protection of young children was applied to the derivation of $PSoQGs_{HH}$ for the protection of adults under the 0.5-IQ point decrement option.

Consequently, for the purpose of derivation of the $PSoQG_{HH}$ for industrial land use, CCME is presenting two options, one using the RSD of 0.5 µg/kg bw/d for the protection of adults, including women of childbearing age to target a 1 IQ point decrement and one using a 50% adjustment factor applied to the equation in order to target a 0.5-IQ point decrement.

6. DERIVATION OF PROVISIONAL HUMAN HEALTH SOIL QUALITY GUIDELINES

The information presented in this chapter includes the revisions made to the human health soil quality guidelines for lead released in 1999 (CCME 1999*a*).

6.1 Protocol

Human health soil quality guidelines provide concentrations of substances in soil below which no appreciable risks to human health are expected. To derive a quantitative guideline, it is necessary to define one or more scenarios by which exposure will occur. According to the Protocol, human health soil quality guidelines are defined for four land uses: agricultural, residential/parkland, commercial and industrial. Since lead is treated as a non-threshold substance, an estimated daily intake (EDI) was not required to calculate the $PSoQG_{HH}$.

During the development of the $PSoQG_{HH}$, alternative approaches to that used by EFSA (2013) were evaluated for estimating lead exposures in children. Examples include biokinetic modelling approaches, which involve transforming media concentrations or exposure doses to BLLs for the purpose of deriving a lead $PSoQG_{HH}$. Peer-reviewed literature and studies by government agencies compared various models to determine their power to predict BLLs as a function of lead intakes and media concentrations. A summary of these efforts was provided by Equilibrium Environmental Inc. (2008*b*) and focused on three of the multi-compartmental models: the US EPA's Integrated Exposure Uptake Biokinetic Model (IEUBK), O'Flaherty (1993; 1998), and Leggett (1993). Since this type of approach required considerable deviation from the Protocol and would represent a novel approach difficult to replicate or to modify for the purposes of deriving site-specific limits, it was not selected for final guideline development.

In the case of lead, CCME developed human health soil quality guidelines for two key age groups: the most sensitive receptors present at residential/parkland, agricultural and commercial sites were toddlers, and at industrial sites, women of childbearing age. A threshold for effects has not been determined for non-cancer effects for lead (ATSDR 2007*a*; CalOEHHA 2007; EFSA 2013; HC 2013*a*; WHO/JECFA 2011), which has implications for the approach used to derive soil quality guidelines (see Wilson and Richardson 2013). EFSA (2013) identifies an estimated exposure level associated with a 1-IQ point decrement, which is considered to be significant on a population level, while WHO/JECFA (2011) provide a range of estimated exposure levels associated with 0.5-IQ to 1-IQ point decrements. CCME has chosen to provide $PSoQGs_{HH}$ based on RSDs associated with both 0.5- and 1-IQ point decrement endpoints for land uses where toddlers are the most sensitive receptors. Adults are assumed to absorb lead at a rate of 40% of the rate that children do (US EPA 2003). Nevertheless, there is some concern that pregnant women may absorb lead at an increased rate compared to non-pregnant women (Franklin *et al*. 1997). Since neither EFSA (2013) nor WHO/JECFA (2011) specifically addressed this concern, CCME has adopted a conservative approach and apply the RSDs associated with IQ decrements in toddlers and children to adult exposures for industrial land uses.

Therefore, CCME used a non-threshold approach, with some modifications to the Protocol, to derive the $PSoQG_{HH}$. Specifically, CCME developed guideline values for two end points (RSDs): targeting a 0.5-IQ point decrement and a 1.0 IQ point decrement in toddlers and adults (including pregnant and breastfeeding women). These risk-specific levels were developed to enable individual jurisdictions to determine their science policy positions for the basis of the $PSoQG_{HH}$.

6.2 Additional Considerations for Lead Provisional Soil Quality Guidelines for Human Health Development

For lead, the following additional considerations must be addressed as part of the derivation process for $PSoQGs_{HH}$:

1. The toxicological data do not currently provide an identified threshold for health effects for lead. As a result, the lead RSDs derived for adults and children are associated with the potential for health risks on a population level. According to the Protocol, SoQGs are typically conservative and are intended to be protective of the majority of individuals in an exposed population. In the case of lead, it is not possible to identify such a low level of effects on an individual basis due to the influence of confounding variables and the related variability in IQ test results. The TRVs are considered provisional (Section 5.7) since they are based on a novel approach, using BLL and a non-threshold approach for a non-carcinogenic substance, given that a threshold for the health effects has not yet been identified. Thus, the lead Soil Quality Guidelines derived herein are considered provisional ($PSoQGs_{HH}$).

2. Since a threshold for effects cannot be determined for lead, it is widely treated as a non-threshold substance (CalEPA 2009; EFSA 2013; HC 2013*a*; US EPA 2006*a*; WHO/JECFA 2011). The exposure limits used for the current guideline calculations correspond to an incremental increase above background (Section 5.7), based on data that failed to identify

an observable effects threshold. Therefore, the non-threshold approach to calculate the lead $PSoQG_{HH}$ was identified as the preferred strategy.

3. Since lead is a non-threshold non-carcinogen, the SQG considers the incremental effect above background (similar to an incremental lifetime cancer risk used for a non-threshold carcinogen). For this reason, background exposure through commercial foods, municipal drinking water, ambient air, consumer products are not considered.

4. In consideration of the two RSD levels (targeting decrements of 1- and 0.5-IQ points), separate SQG values have been developed to enable individual jurisdictions to determine their science policy positions with regards to the $PSoQG_{HH}$. The 1-IQ point decrement reflects EFSA's 0.5 µg/kg bw/d intake that is associated with the $BMDL_{01}$ for that effect at a population level. The 0.5 IQ point decrement is based on application of a 50% adjustment factor to target the upper end of the range of "negligible" impacts, as identified by WHO/JECFA (2011).

5. Jurisdictions may differ in selection of a PSoQG based on a 0.5- or 1-IQ point decrement. CCME recommends that jurisdictions select a 0.5-IQ point decrement to allow for background exposure to all sources of lead, including, but not limited to the diet, drinking water, and air. Jurisdictions that adopt a PSoQG based on a target 1-IQ point decrement may also want to apply the PSoQG based on a 0.5-IQ point decrement in situation where soils may also result in contamination of water used as drinking water and/or consumption of food grown on-site.

6. Due to the differing sensitivities of adults and children, different BLL-based lead RSDs (toddler and adult) are available (see Section 5.7). However, because the adult RSD may not provide adequate protection to both developing fetuses and women of childbearing age, the toddler RSDs were used to develop all guidelines.

7. Typically, the non-threshold calculation approach involves only an adult receptor (see CCME 2006), as the cancer endpoint requires a long latency period. Due to the inherent sensitivities of children to lead and the lack of prolonged latency with respect to the manifestation of effects, CCME determined that both adults and toddlers would be included in the $PSoQG_{HH}$ calculation for all land uses to ensure a conservative approach was implemented (Section 5.7). The toddler RSDs were used to develop all guidelines because the adult TRV may not provide adequate protection to developing fetuses (e.g., via the exposure of women of childbearing age). Women of childbearing age commonly use industrial land sites, and their presence is therefore assumed for this generic land use.

8. The off-site migration check for industrial land uses was considered protective of the potential receptors near industrial sites, where water-borne and windblown soil may affect soil concentrations on nearby agricultural and residential lands. However, the off-site migration check does not address situations where soils from shoes or clothes may affect indoor settled house dust. In cases where this may be a concern, a site-specific assessment will be necessary.

6.3 Summary of Toxicological Reference Values for Human Receptors

6.3.1 Infants, Toddlers, Children, and Adolescents

Based on the available data, CCME is presenting two options, one using the EFSA (2013) intake of 0.5 µg/kg bw/d as an RSD targeting a 1-IQ point decrement and one using the RSD with a 50% adjustment factor to target a 0.5-IQ point decrement for the purposes of $PSoQG_{HH}$ development (Section 5.7). The key study used to derive the child RSD for lead was the pooled analysis by Lanphear *et al.* (2005). This study included a large number of diverse subjects with a sufficient number of pre-school and school-age children with BLLs ≤ 10 µg/dL to give it sufficient statistical power to describe the relationship between blood lead and cognitive function.

6.3.2 Adults

To evaluate adults, SBP was selected as the critical endpoint evaluated by WHO/JECFA (2011), while SBP and renal effects were selected as the critical endpoints by EFSA (2013). However, neither WHO/JECFA (2011) nor EFSA (2013) provide specific conclusions on whether the proposed intake rates for adults would also be protective of developing fetuses. Therefore, CCME adopted the EFSA (2013) 0.5 µg/kg bw/d intake rate as an RSD targeting a 1-IQ point decrement and applied a 50% adjustment factor to also target a 0.5-IQ point decrement to generate two options for $PSoQGs_{HH}$ for all land uses to protect all age groups including women of childbearing age.

6.4 Bioavailability (Relative Absorption Factors)

Potential health risks posed by inhaling, ingesting or having dermal contact with lead in soil depend on the absorbed dose that could subsequently be delivered to target tissues and elicit an adverse effect. The terms relative bioavailability or RAF are conceptually similar within the context of risk assessment. However, they are used to express differences in the absorption of a substance between two media of exposure, which may not necessarily include the media upon which a critical toxicological study is based.

Frequently, for the purposes of $PSoQG_{HH}$ development, it is assumed the bioavailability of a substance in soil and from different exposure routes is equal to the bioavailability calculated from the TRV study. This assumption is due in part to the lack of adequate data (RAF = 1 or 100% is assumed for soil ingestion and re-suspended dust inhalation). Variable RAFs are often applied for dermal contact, since few exposure limits are based on dermal exposure.

Dodd et al. (2017) reported a range of lead bioaccessibility in Canadian background soils (20 to 93%), based on 114 samples from ten locations across Canada. These results illustrate the high variability of lead bioaccessibility in Canadian soils. Both US EPA and HC recommend that site-specific testing be done when feasible to increase the accuracy (reduce uncertainty) of exposure estimates and risk calculations because site-specific factors can affect the bioavailability of metals in soils from one site to another or even within a site, depending on contaminant source, form, soil matrix, and other site-specific factors (HC 2017*b*; US EPA 1994*b*). The US EPA (2007*a*) regression equation for lead can be used to estimate an oral RAF based on site-specific

bioaccessibility test results. For the purpose of developing the $PSoQGs_{HH}$, a default oral RAF of 1 was used. For dermal exposures, CCME adopted a dermal RAF for lead of 0.6%, based on the analysis performed by Moore *et al.* (1980). For inhalation exposures, the RAF was assumed to be 100%. See Section 5.3.1 for more information on the various media and routes of exposure considered in the absorption of lead. Various *in vitro* and *in vivo* methods are also available to estimate the relative and absolute bioavailability of lead bound to soil via the oral route at a particular site, allowing for the site-specific modification of guidelines. CCME recommends that practitioners intending to measure site-specific bioavailability values consult appropriate provincial and federal agencies for guidance.

7. CALCULATION OF SOIL QUALITY GUIDELINES FOR HUMAN HEALTH

Different algorithms and assumptions are applied to calculate $PSoQGs_{HH}$ for agricultural, residential/parkland, commercial and industrial land uses. The key receptor of concern for agricultural and residential/parkland uses is the toddler. For commercial land use, the toddler is considered the most sensitive receptor, and for industrial land use, only the adult is evaluated.

7.1 Agricultural and Residential/Park Land Uses

To determine agricultural and residential/parkland soil guidelines for lead, toddlers are considered the critical receptors due to their larger exposure per unit mass compared to children, adolescents and adults. In accordance with CCME guideline derivation procedures (CCME 2006), CCME derived a soil quality guideline for three soil exposure pathways (ingestion, inhalation and dermal). As discussed in Section 6.2, lead was considered a non-threshold substance due to its mechanism of action. As a result, the $PSoQG_{DH}$ for lead was calculated using the Protocol recommended equation for non-threshold substances (Equation 1).

Equation 1.

$$PSoQG_{DH} = \frac{RSD \times BW}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S) \times ET_2] \times ET_1} + BSC$$

Where:

$PSoQG_{DH}$ = Provisional direct health-based soil quality guideline (mg/kg)
RSD = Risk-specific dose (toddler: 0.00025 or 0.0005 mg/kg-d [EFSA 2013 potency estimate], with or without the 50% adjustment to target 0.5- and 1-IQ point decrements, respectively)
BW = Body weight (toddler: 16.5 kg; CCME 2006)
BSC = Background lead concentration in soil (10 mg/kg)
AF_G = Relative absorption factor for gut (1; based on US EPA [2007*b*] equation [see Section 2.3.1] and assumption of 100% *in vitro* bioaccessibility assay)
AF_S = Relative absorption factor for skin (0.006)
AF_L = Relative absorption factor for lung (1.0)

SIR = Soil ingestion rate (toddler: 8.0×10^{-5} kg/day; CCME 2006)
SR = Soil dermal contact rate (toddler: 6.9×10^{-5} kg/day; CCME 2006)
IR_S = Soil inhalation rate (toddler: 6.3×10^{-9} kg/day; Allan *et al*. 2008)
ET_1 = Exposure term 1 (unitless) for all pathways: days/7 days × weeks/52 weeks at the site
ET_2 = Exposure term 2 (unitless) for the inhalation pathway: hours/24 hours at the site

For the terms ET_1 and ET_2, CCME assumed the following while calculating the $PSoQG_{DH}$ for agricultural and residential/parkland land uses:

ET_1 = exposure term 1 (unitless) for all pathways = 1.0 (i.e., (7/7 days/week × 52/52 weeks/year at the site; CCME 2006)
ET_2 = exposure term 2 (unitless) for the inhalation pathway = hours/day at the site = 1.0 (i.e., 24/24 hours/day assumed at the site (CCME 2006)

Based on the above equation and the RSDs presented in Section 5.7, CCME calculated lead $PSoQGs_{DH}$ for the toddler of 61 and 113 mg/kg for residential land use, for the protection of 0.5- and 1-IQ point decrements, respectively.

7.2 Commercial Land Use

Commercial land use sites are generically defined as sites at which commercial activities predominate, such as a shopping mall or commercial building (where a daycare may operate), but at which no manufacturing activities or residences are present. Therefore, both toddlers and adults may be present, and as a result, the lead $PSoQG_{HH}$ was calculated for toddlers (as the most sensitive receptors).

The commercial land use calculation is nearly the same as the agricultural and residential/parkland calculations, with slight differences. CCME used Equation 1 to derive the $PSoQG_{DH}$ using the following exposure terms:

ET_1 = exposure term (unitless) = 0.71 (5/7 days/week; based partially on CCME 2006)
ET_2 = 0.42 (10/24 hours/day) for the inhalation pathway, due to the reduced amount of time the receptor spends on a commercial site (CCME 2006).

An exposure amortization of 0.71 was considered to be appropriate for ET_1 (i.e., five days/week, 52 weeks/year) for commercial land use. Lead has a relatively long half-life in blood (in the range of 30 to 40 days), so the amortization of five days over seven days is expected to have a very minor influence on the steady-state concentrations (e.g., due to the long half-life, a person exposed to lead at a rate of 0.1 µg/kg bw/day for seven days/week would have a very similar BLL to someone exposed to 0.14 µg/kg bw/day for five days/week). Nevertheless, due to concern for developmental and neurological effects that may be overlooked if even greater amortization was incorporated, CCME did not consider it appropriate to further amortize ET_1 on a number-of-weeks-per-year basis (i.e., 52 weeks per year was assumed, rather than 48 weeks/year per the Protocol). To evaluate inhalation intake, CCME considered a value of 0.42 to be appropriate for ET_2 (i.e., 10 hours/day) for commercial land uses to derive a lead $PSoQG_{DH}$.

Based on Equation 1 and the RSDs presented in Section 5.7, CCME calculated lead $PSoQGs_{DH}$ for the toddler life stage of 82 mg/kg and 154 mg/kg for commercial land use, for the protection of 0.5- and 1-IQ point decrements, respectively.

7.3 Industrial Land Use

In an industrial scenario, the primary route of exposure is occupational, and it is assumed that only adults are present on site. Therefore, the adult receptor/life stage is used for SoQG development for industrial land use. Nevertheless, women of childbearing age and pregnant women may also be present at industrial sites. Consequently, the $PSoQGs_{DH}$ were derived using both the RSD protective of 0.5- and 1.0-IQ point decrements (0.00025 mg/kg bw/day and 0.0005 mg/kg bw/day) to account for the possible presence pregnant and nursing women, along with adult soil contact rates. Specifically, CCME estimated the $PSoQGs_{DH}$ using Equation 1 with the following adult-specific values for each RSD:

BW = Body weight (adult: 70.7 kg; CCME 2006)
SIR = Soil ingestion rate (adult: 2.0×10^{-5} kg/day; CCME 2006)
SR = Soil dermal contact rate (adult: 1.7×10^{-4} kg/day; CCME 2006)
IR_S = Soil inhalation rate (adult: 1.3×10^{-8} kg/day; Allan *et al*. 2008).

Exposure at an industrial land use site is assumed to be 10 hours/day, five days/week and 52 weeks/year, and thus the ET_1 and ET_2 values are the same as for commercial land use. Based on Equation 1 presented in Section 7.1, the lead $PSoQGs_{DH}$ for industrial land use are calculated to be 1,194 mg/kg and 2,378 mg/kg, for the protection of a 0.5 and 1 IQ point decrements, respectively. However, the calculated off-site migration check values (see section 7.4) are lower than the $PSoQGs_{DH}$ calculated for the industrial land use so the $PSoQGs_{DH}$ are adjusted based on the off-site migration check and set at 743 mg/kg and 1,477 mg/kg for the protection of a 0.5- and a 1-IQ point decrement, respectively.

7.4 Off-site Migration Check

When deriving $SoQGs_{HH}$ for commercial and industrial land uses, CCME's exposure scenarios only consider on-site exposure. Soil containing elevated levels of lead can be transferred from one property to another by both wind and water erosion (CCME 2006). The human health soil quality guideline for off-site migration ($SoQG_{OM\text{-}HH}$) refers to the concentration in soil eroded from the site that could raise the lead concentration in the receiving soil to the level of the agricultural guideline within a specific time frame. The $PSoQG_{OM\text{-}HH}$ is derived as follows:

Equation 2.

$$PSoQG_{OM-HH} = 14.3 \times PSoQG_{A-HH} - 13.3 \times BSC$$

Where:

$PSoQG_{OM\text{-}HH}$= Human health-based soil quality guideline for off-site migration (mg/kg)

$PSoQG_{A\text{-}HH}$ = Human health-based soil quality guideline for agricultural land use (61 mg/kg or 133 mg/kg for the protection of a 0.5- and 1-IQ point decrement, respectively)
BSC = Background concentration of lead in the receiving soil (10 mg/kg [till]).

Using the equations and assumptions above, CCME estimates the concentration of lead in the eroded soil to be 743 mg/kg and 1,477 mg/kg, for the protection of a 0.5 and 1 IQ point decrements, respectively. For the protection of a 0.5-IQ point decrement, the calculated $PSoQG_{OM\text{-}HH}$ of 743 mg/kg is less than the $PSoQG_{HH}$ of 1,194 mg/kg. For the protection of a 1-IQ point decrement, the $PSoQG_{OM\text{-}HH}$ of 1,477 mg/kg is less than the $PSoQG_{HH}$ of 2,378 mg/kg. Therefore, the industrial land use $PSoQGs_{HH}$ were adjusted based on this check and set at 743 mg/kg and 1,477 mg/kg for the protection of a 0.5- and a 1-IQ point decrement, respectively.

7.5 Guideline for Protection of Groundwater

No guideline for the protection of groundwater from soil leaching was derived for lead, due to restrictions on the mathematical model when applied to metals (Nason 1996).

7.6 Produce, Meat and Milk Check

A produce, meat and milk check was not performed due to insufficient data.

8. RECOMMENDED CANADIAN SOIL QUALITY GUIDELINES

According to the Protocol, both $SoQGs_E$ and $SoQGs_{HH}$ are developed for four land uses: agricultural, residential/parkland, commercial and industrial. CCME recommends the lowest value generated by two approaches for each of the four land uses as the CSoQG. A threshold for the human health effects of lead has not yet been identified. The TRV is provisional, given that they are based on a novel approach, using BLL and a non-threshold approach for a non-carcinogenic substance. The $SoQGs_{HH}$ are therefore also considered provisional. $SoQGs_E$ are not addressed in this document; however, those derived in 1999 (EC 1999*a*) are retained herein for the selection of the recommended CSoQGs as presented in Table 3.

Table 3. Soil quality guidelines for lead (mg/kg)

	Land use			
	Agricultural	**Residential/ parkland**	**Commercial**	**Industrial**
Provisional Soil quality guideline (PSoQG) -0.5-IQ pt decrement [a, c]	61	61	82	600
Provisional Soil quality guideline (PSoQG) -1-IQ pt decrement [a, d]	70	113	154	600
Human health guidelines/check values				
$PSoQG_{HH}$ -0.5-IQ pt decrement [b, c]	61	61	82	743
$PSoQG_{HH}$ -1-IQ pt decrement [b, d]	113	113	154	1,477
Direct contact ($PSoQG_{DH}$) – 0.5-IQ pt	61	61	82	1,194
Direct contact ($PSoQG_{DH}$) – 1-IQ pt	113	113	154	2,378
Protection of indoor air quality ($SoQG_{IAQ}$)[e]	NC	NC	NC	NC
Protection of potable water ($SoQG_{PW}$)[f]	NC	NC	NC	NC
Off-site migration check ($PSoQG_{OM-HH}$) – 0.5-IQ pt	–	–	743	743
Off-site migration check ($PSoQG_{OM-HH}$) – 1-IQ pt	–	–	1,477	1,477
Produce, meat and milk check ($SoQG_{FI}$)[g]	NC	NC	–	–
Environmental health guidelines/check values (EC 1999)				
$SoQG_E$[h]	70	300	600	600
Soil contact ($SoQG_{SC}$)	300	300	600	600
Soil and food ingestion ($SoQG_I$)	70	–	–	–
Protection of freshwater life ($SoQG_{FL}$)[f]	NC	NC	NC	NC
Livestock watering ($SoQG_{LW}$)	–	–	–	–
Irrigation water ($SoQG_{IR}$)	–	–	–	–
Nutrient and energy cycling check ($SoQG_{NEC}$)	723	723	834	834
Off-site migration check ($SoQG_{OM-E}$)	–	–	870	870
Superseded soil quality guidelines (CCME 1999)	–	140	260	600
Superseded interim soil quality criteria (CCME 1991)	375	500	1,000	1,000

Notes: NC = not calculated; $PSoQG_{HH}$ = provisional soil quality guideline for human health; $SoQG_E$ = environmental soil quality guideline; a dash indicates a guideline/check value that is not part of the exposure scenario for that land use and therefore is not calculated. Soil guidelines and the data used to calculate them are, by convention, always expressed on a dry weight basis to allow the data to be standardized. In case of doubt and if the scientific criteria document does not specify whether wet or dry weight is used, readers are advised to check the references provided.

[a] Data are sufficient and adequate to calculate a $PSoQG_{HH}$ and a $SoQG_E$. The lower of the SoQGs becomes the soil quality guideline for this land use.

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

[c] CCME recommends that the $PSoQG_{HH}$ for a 0.5-IQ point decrement be used when soil and additional site-related media contain elevated concentrations of Pb (e.g., groundwater, food grown on site, etc.) to account for additional sources of elevated exposure and/or according to jurisdictional policy.

[d] CCME recommends that the $PSoQG_{HH}$ for a 1-IQ point decrement be used when soil is the only site-related media with elevated concentrations of Pb. Where additional site-related contaminated media are expected to contribute to exposures (e.g., groundwater,

food grown on site, etc.), CCME recommends using the 0.5-IQ point decrement to account for additional sources of elevated exposure. When using the PSoQGHH based on 1-IQ point decrement, the environmental site investigation report should include information on all media that may be affected above background and fate and transport information.

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

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

[g] Not calculated due to high site-related variability in soil parameters and intake rates. Where these pathways are complete, assess on a site-specific basis.

[h] The SoQGE is the lowest of the ecological guidelines and check values.

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APPENDIX 1. LEAD CONCENTRATIONS IN THE CANADIAN ENVIRONMENT

Air

Location	Year	Mean concentration (SD) µg/m^3	Range µg/m^3	Comments	Reference
8 Canadian cities	1986–96	0.0405	-	$PM_{2.5}$ only	Burnett *et al.* 2000
Ontario: Windsor	2004	124 µg/g	46–169 µg/g	$PM_{2.5}$	Rasmussen *et al*. 2007
	2005	0.0072 (0.0029)	0.0031–0.013	$PM_{2.5}$ 24hr sample; ICP-MS	Niu *et al*. 2010
	2005	0.0036 (0.0008)	0.0023–0.0059	$PM_{2.5}$ 2wk sample; ICP-MS	Niu *et al*. 2010
Ontario: Ottawa					
Rural	-	0.0038	0.0014–0.0063	$PM_{2.5}$	Rasmussen *et al*. 2006
Urban	-	0.0033	0.0024–0.004		
Edmonton	2001–2002				
Northwest		0.002	0.0167 (max)		
AADAC site		0.002	0.0069 (max)	$PM_{2.5}$	AENV 2003
Polytubes site		0.0019	0.0096 (max)		
Unisource site		0.0014	0.0068 (max)		
British Columbia					
Castlegar	-	0.1305	4.355 (max)	$PM_{2.5}$	Biometrics 2000
Trail: Butler Park	-	0.3085	2.185 (max)		
Burnaby	-	0.01225	0.0025–0.035	-	Brewer and Belzer 2001
Golden	-	0.29	0.02–1.55	$PM_{2.5}$	Evans and Jeong 1997

Location	Year	Mean concentration (SD) µg/m^3	Range µg/m^3	Comments	Reference
Manitoba: Transcona	2003–2005	0.0022	0.0003–0.0149	PM_{10}	Van Dusen 2006
		0.0014	0.0002–0.0114	$PM_{2.5}$	
Québec	-	0.0025–0.005	-		Bisson 1997
United Sates: New York–New Jersey Harbor, NY/NJ	1998–1999				
Liberty Science Center		0.0079 (0.0054)	-	$PM_{2.5}$	Gao *et al*. 2002
New Brunswick		0.0066 (0.0065)	-		
Sandy Hook		0.0049 (0.0036)	-		
United Sates: Minneapolis/St. Paul, MN	1999	0.002	-	$PM_{2.5}$	Adgate *et al.* 2007
United States: EPA Region V	1995–1997	0.0112	-	Size fraction not reported	Clayton *et al.* 1999

Indoor air

Location	Year	Mean concentration (SD) ng/m^3	Range ng/m^3	Comments	Reference
Alberta: High level	1997	0.59		2 houses sampled per day over 10 days ($PM_{2.5}$)	Alberta Health 1998
Ontario	2004	1189 µg/g	262–2651 µg/g	$PM_{2.5}$	Rasmussen *et al*. 2007
Windsor: offices (smoking possible)	-	9.0	0.7–47.1	Fraction size not reported	Bell *et al.* 1994
Windsor: non-smoking	-	7.8	2.1–23.1		
Windsor: smoking	-	12.6	2.4–47.0		
Windsor: smoking + non-smoking	-	9.7	2.1–47.0		
Ottawa: rural		2.3 (median)		Median from non-smoking homes; $PM_{2.5}$	Rasmussen *et al*. 2006
Ottawa: urban		1.5			
United States: Baltimore, MD	1998	1.03		Median indoor central aerosol ($PM_{2.5}$)	Graney *et al.* 2004
Minneapolis/St. Paul, MN	1999	3.4		$PM_{2.5}$	Adgate *et al.* 2007

Soil

Location	Year	Soil type	Sample depth	Mean concentration (SD) mg/kg	Range mg/kg	Comments	Reference
Canada	2008–2009	Surface	0–5 cm	21.85 (21.18)	1.61–227.84	SK, QC, ON, NS, NL, MB, BC, AB	Friske *et al*. 2014
	-	Glacial tills		9.68 (7.93)	2–152	n = 7398	Grunsky 2010
	-			19	2.3–250	Background samples	Sheppard *et al*. 2007
Canada	-	Uncultivated soils	A, B & C horizons	20 (NR)	5–50	Uncontaminated, remote from ore bodies	McKeague and Wolynetz 1980
Appalachian Region				21 (NR)			
Canadian Shield				21 (NR)			
St. Lawrence Lowlands				25 (NR)			
Interior Plains				15 (NR)			
Cordilleran Region				16 (NR)			
Eastern Canada	-	-	-	22.93 (9.44)	2.40–57.80		Levesque and Mathur 1986
AB: Cold Lake	2004	-	-	13 (NR)	3–37		DRDC 2004
BC: Trail	1994–1999	Residential	Top 2–3 cm	750	-		Hilts 2003
BC: Cariboo Region		-	-	9.5		Regional background (95th percentile)	BC MOE 2010
BC: Kootenay Region		-	-	75			
BC: Lower Mainland Region		-	-	60			
BC: Omineca Peace Region		-	-	35			
BC: Skeena Region		-	-	15			
BC: Thompson Nicola Okanagan		-	-	15			
BC: Vancouver Island Region		-	-	30			
BC: Vancouver		-	-	300			
BC: Victoria	2003	-	-	90	-		Bowman *et al*. 2003
MB: Flin Flon	2006	-	-	196.4	5–1447		Manitoba Conservation 2007
MB: Cranberry Portage		-	-	21.9	-		
MB: Bakers Narrows		-	-	5.3	-		
MB: Creighton	2006	-	Top 2.5 cm	92.5	6.3–250		

Location	Year	Soil type	Sample depth	Mean concentration (SD) mg/kg	Range mg/kg	Comments	Reference
MB: Flin Flon		-		196.4	5.0–1446.7	Soil affected by point source Pb emissions	Manitoba Conservation 2007
NB: St. John	2003	-	-	203 (median)		Residential soil along foundations and road sides	Bell *et al*. 2010
NL: Buchans	2009	Surface soil	-	773	27–4800	Background residential	Conestoga-Rovers and Associates 2010
NS: Sydney	2004	Residential area near contamination source	Top 5 cm	297	52–1700	Community surrounding Sydney Tar Ponds	Lambert and Lane 2004
	-	-	Top 5 cm	150	15–3800	Urban reference area	JDAC Environment Limited 2001
NS: Halifax	2010	-	0–10 cm	109	10–767	Open spaces, community, backyard, and raised-bed gardens	Heidary-Monfard 2011
NU: Iqaluit	2007	Urban brownfield	10 cm		17–45	Residential neighbourhood	Laird 2010
NT: Lac de Gras	2004–2005	Loamy sand to sandy loam	0–10 & 20–30 cm	-	< 5–10	Randomly located samples of till, topsoil	Naeth *et al*. 2006
ON: Toronto	2008	Urban contaminated	0–15 & 15–30 cm	38	9–210	Lakeshore Boulevard East between Don Roadway and Leslie St	Aqua Terre Solutions Inc. 2009
ON: Sudbury	2004	Contaminated	0–5 cm	50	3.5–194	Sudbury Area	Centre for Environmental Monitoring 2004
ON: Ottawa	2000	Garden soils	Top 5 cm	64.69	15.6–547.44	Private dwellings built between 1893 & 1987. Composite samples	Rasmussen *et al*. 2001
ON: Guelph	1990–2007	Residential neighbourhood	-	142	5–420	W of Victoria Rd and N of the Eramosa River	Wellington-Dufferin-Guelph Public Health 2007
ON: Lanark County	2003	-	-	24	-		Rasmussen 2004
QC: St. Clair Watershed	2005		10–19.75 cm	18	15–29	Natural background levels	Ndzangou *et al*. 2005
QC: Massawippi and Saint-François Rivers	2010	Alluvial soils (riparian zone)	various	48.0	13.2–149		Saint-Laurent *et al*. 2010

Location	Year	Soil type	Sample depth	Mean concentration (SD) mg/kg	Range mg/kg	Comments	Reference
		and agricultural areas					
YT: Carmacks	2009	-	10–90 cm	10.8	1–126	Paleozoic, Mesozoic & Tertiary metamorphic, igneous & sedimentary rocks	Wesgold Minerals Inc 2010
United States							
ID: Northern Idaho	-	Yard soil	0–2.54 cm	79 (88)	5–525	Background samples	Spalinger *et al*. 2007
MI: Rouge River Watershed	-	Commercial	< 0.5 m	93 (300)	-		Murray *et al*. 2004
	-	Residential	< 0.5 m	160 (250)	-		
	-	Industrial	< 0.5 m	150 (380)	-		
	-	Commercial	0.5–10 m	20 (60)	-		
	-	Residential	0.5–10 m	34 (77)	-		
	-	Industrial	0.5–10 m	39 (72)	-		
SC: South Carolina	-	Residential and industrial areas	0–5 cm	12	2.1–53		Aelion *et al*. 2008
		Forests, residential, commercial and industrial areas		30	6.5–200		
	-	Rural area		12	2.1–53		Aelion *et al*. 2009
	-	Rural area		17	1.6–140		
	-	Urban area		45	2.4–288		
NJ: Jersey City		Yard soil	Top 5 cm	888 (arithmetic); (SD 734); 538 (geometric)	70–2080	Geometric mean provided in journal; other stats calculated from data presented in Adgate 1998*b*	Adgate *et al*. 1998*b*
United States: overall	-	Agricultural soils	Top 50 cm	12.3	0.5–135		Holmgren *et al*. 1993

Dust

Location	Year	Sample location	Mean concentration (SD) mg/kg	Range mg/kg	Comments	Reference
Canada: 13 cities	2007–2010	House dust: Urban background	58		Geomean	Rasmussen *et al*. 2011
		Elevated	447			
		Anomalous	1,730			
Canada: 13 cities	2007–2010	House dust	119	14.2–7,800	Geomean	Rasmussen *et al*. 2013
Ontario: Ottawa	1993	Indoor dust	405.6	50.2–3,226	50 homes from 10 neighbourhoods	Rasmussen *et al*. 2001
BC: Trail	1999	House dust	583	-	Geomean	Hilts 2003
United States	-	All surfaces	493.1	1.1–30,580	Geomean	Adgate *et al*. 1995
	-	Surface dust	516.2	3.7–30 580		
	-	Windowsill dust	1,031.9	1.1–21,120		
	-	Dust plate	147.1	2.9–1,497		Adgate *et al.* 1998*a*
	-	Carpet dust	104.1	3.9–2,682		
	-	Windowsill and floor dust	1,200	47–18,600	Arithmetic mean	Adgate *et al*. 1998*b*
United States**:** Region V	-	Surface dust	463.1	-	Arithmetic mean	Clayton *et al.* 1999
	-	Windowsill	954.1	-		
United States: Northern Idaho	1998	House dust	547	-	Geomean	von Lindern *et al*. 2003
	1999	Floor mat dust	162 (229)	NS-1,280	Arithmetic mean	Spalinger *et al*. 2007
		Vacuum dust	187 (192)	NS-830		
United States: New Jersey	1992–94	Indoor dust PM_{10}	1,133 (119)	-	Concentrations converted from mass %.	Adgate *et al.* 1998*a*
Australia: Sydney	1999	Indoor dust	389 (1,890)	18.2–16,600	82 homes from 6 suburbs	Chattopadhyay *et al.* 2003
Sydney	1997 & 1999	Indoor dust	1,830 (703)	480–2,594	< 500 m from industrial site (n = 10)	Davis and Gulson 2005
			1,462 (1,514)	799–6,997	500–1,500 m from industrial site (n = 19)	
			604 (358)	105–1,150	> 1,500 m from industrial site (n = 8)	

Location	Year	Sample location	Mean concentration (SD) mg/kg	Range mg/kg	Comments	Reference
New Zealand: Christchurch	1987	Indoor dust	724 (577)	101–3,510		Kim and Fergusson 1993
United Kingdom	2005	Indoor dust	181 (104)	56.8–358		Turner and Simmonds 2006
Germany	1990–1992	Indoor dust	56.9	NS-37,000		Seifert *et al.* 2000
Poland: Warsaw	1997	Indoor dust	144 (66) 169 (66) 209 (80)	64–318 80–318 91–334	63–125µm 32–63µm 0–32µm	Lisiewicz *et al.* 2000
Bahrain	-	Indoor dust	517 (183)	211–1,010		Madany *et al.* 1994
	-	Outdoor dust	742 (236)	328–4,627		

NS = not specified

Sediment

Location	Year	Mean concentration mg/kg dw	Range mg/kg dw	Comments	Reference
Canada: Lake	-	6	-		CCME 1999*b*
Stream		12.7			
Marine	-	-	7–23.4		EC 1998
Surficial sediments near source of contamination	-	3,000 (max mean)		Freshwater lakes & rivers near manufacturing plants	CCME 1999*b*
	-	15,400 (max mean)		Marine harbours receiving industrial & sewage inputs	
Yukon	-	56.22	-		Gamberg *et al.* 2005
Canadian Arctic and Subarctic					Evans *et al*. 2005
Western streams		21.8±20.0		n = 4	
Eastern streams		16.2±10.05		n = 6	
Lake, offshore of streams		21.6±8.1		n = 6	
Lake outflow, 3 m		13.6		n = 1	
BC: Fraser River Basin	1993–1994	8.4–30.9	-	Background (pre-1900)	Gallagher *et al.* 2004
ON: Great Lakes	2001–2002	27.9–47.5	< 1–287		Gewurtz *et al.* 2008
Lake Erie	1997–1998	18.2	-	Background	Marvin *et al.* 2004
Lake Ontario	-	15.0	-		

Location	Year	Mean concentration mg/kg dw	Range mg/kg dw	Comments	Reference
Lake Simcoe	2008	-	40–81	2000–present	Landre *et al*. 2011
		-	10–15	Pre-1900s	
ON: Lake Erie	2001	-	16.1–117		Marvin *et al*. 2007
ON: 12 lakes	1998	-	0.06–20.7	Near shore sediment	Shuhaimi-Othman *et al*. 2006
ON: Killarney Park	-	-	Max range: 65–255	Remote area	Belzile *et al*. 2004
ON: Sudbury	2001	5.9–204.7	-	Lake samples	Pyle *et al*. 2005
QC: Montreal (St. Lawrence fluvial lakes)	2004–2005	-	6–140		Desrosiers *et al*. 2008
Bay of Fundy	1997–2002	66	8–26		Hung and Chmura 2007

Drinking water

Location	Year	Mean concentration µg/L	Range µg/L	Comments	Reference
Canada-wide	Summer 2009–2010	1.27	0.5–24	Drinking water distribution system samples. Median concentrations	HC 2014
	Winter 2009–2010	0.9	< 0.5–8.2		
ON, NL and SK	-	1.12 (3.64)	-	n = 14,408	SENES 2010
ON	1998–2004	0.73 (2.19)	-	n = 3,350	
SK	-	1.12 (4.61)	-	n = 2,524	
NL	-	1.27 (3.76)	-	n = 8,534	
BC: Victoria	2008–2009	0.3 (1.6)	< 0.2–NA	Exiting treatment facility	CRD 2010
YT	2005–2010		< 0.1–7.6	Residential	Yukon Environmental Health Services 2011
AB: Calgary	2018	< 0.5	-	Exiting treatment facility	City of Calgary 2019
AB: Edmonton	2019		< 0.2-88.2	Residential	EPCOR 2020
MB: Portage la Prairie	2008–2009	0.7	0.1–36	Residential	Manitoba Conservation and Water Stewardship 2013
MB: Winnipeg	2018		0.05-1.8		City of Winnipeg 2019
SK	2005–2010	6.7	0.1–60	Residential; n = 176	Government of Saskatchewan 2011
SK	2009	6.7 (median)	< 0.1–60	Residential; median	Government of Saskatchewan 2011

Location	Year	Mean concentration µg/L	Range µg/L	Comments	Reference
SK: Saskatoon	2019	< 0.1	-		City of Saskatoon 2020
ON	2012		< 0.5–10.8	Residential	OMOE 2012
ON: Hamilton	2019	-	< 0.1-44.4	Residential	City of Hamilton 2019
ON: Waterloo	2019	-	ND-46.1	Residential	City of Waterloo 2009
ON: Barrie	2019	-	<0.02-3.97	Residential	City of Barrie 209
ON: Kingston	2019	-	< 0.02–0.16		City of Kingston 2019
ON: Ottawa	2019	-	< 0.1-2.2	Residential	City of Ottawa 2020
ON: St. Catharines	2019	-	1-43	Residential	City of St. Catherines 2020
ON: Sudbury	2019	-	0.1-0.77		City of Sudbury 2020
ON: Thunder Bay	2019	-	0-271	Residential	City of Thunder Bay 2020
QC	2005–2010	2–5 (annual median)	0.06–530	Residential; n ≥ 13,000	MDDEP 2011
QC: Montreal	2007	14	2.1–114	Residential; median dissolved Pb	Deshommes *et al*. 2010
		0.39	< 0.02–12	Residential; median particulate Pb	
QC: Montreal	2009–2010	1.76	0.06–32.05	Residential; 4 boroughs	INSPQ 2010
QC: Montreal	2016	-	0.79-0.89		City of Montreal 2016
QC	2005–2010	2–5	0.06–530	Residential; geometric means	Province of Québec 2011
PE	2005–2010	-	< 2–335	Private wells; n ≥ 10,000; 88% < MDL	PEI DEEF 2011
PE: Charlottetown	2018	< 2.0	-		City of Charlottetown 2019
NS: Halifax	2008–2010	< 0.5 (median)	-		City of Halifax 2011
NS: Dartmouth	2009–2010	< 0.5 (median)	-		
NL	2019	-	< 1.0–21	Residential	Newfoundland and Labrador 2019

Surface water

Location	Year	Concentration µg/L	Range µg/L	Comments	Reference
Canada: agroecosystems		3	-	River samples	He *et al.* 2005
St. Lawrence River	1987	0.019	0.009–0.035	Average dissolved Pb concentrations between Lake Ontario and Québec City	Lum *et al*. 1991

Location	Year	Concentration µg/L	Range µg/L	Comments	Reference
ON: 12 lakes	1998	-	0.09–0.62	Surface water, affected lakes 6–12 km from smelter	Shuhaimi-Othman *et al.* 2006
	1998		0.06–0.23	Intermediate lakes 32–52 km from smelter	
	1998		0.08–0.14	Control lakes 94–154 km from smelter	
Nova Scotia					Nova Scotia Environment 2010
Kelley River	2005–2008	1	1–1	n = 16; values are ½ DL of 2	
Margaree River	2002–2008	0.9	0.3–1	n = 24	
Pockwock Lake	2002–2008	1	1–1	n = 7; values are ½ DL of 2	
Shelburne River	2002–2008	1	1–1	n = 20; values are ½ DL of 2	
St. Mary's River	2007	1	1–1	n = 6; values are ½ DL of 2	
United States: Surface water	-	-	5–30	≤ 890 µg/L have been reported	ATSDR 2007*a*
River water	-	5	0.6–120.0	Median in natural river water	
Surface water	-	3.9	-	Mean from from 50,000 surface water stations	Eckel and Jacob 1988
VT	1998	0.54	0.03–1.45	Rain samples	Malcolm *et al.* 2003

Commercial foods

Food type	Year	Mean concentration	Range (g/L – water; µg/kg – food)	Comments	Reference
Canada	1995–1996	0.32	0.26–0.97	Retail distilled water	Dabeka *et al*. 2002
		0.69	0.27–10	Retail mineral water	
		0.49	0.27–7.97	Retail spring water	
Canada: Total Diet Studies	2003–2007			Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2010*a*
Dairy products		-	< 0.19–13.5		
Meat and meat products		-	1.21–32.6		
Poultry and poultry products		-	< 0.91–8.34		

Food type	Year	Mean concentration	Range (g/L – water; µg/kg – food)	Comments	Reference
Fish and fish products		-	0.83–20.7		
Soups		-	1.69–6.94		
Cereal and cereal products		-	0.86–15.9		
Vegetable and vegetable products		-	0.54–23.8		
Fruit and fruit products		-	0.55–83.4		
Fats and oils		-	< 0.18–4.2		
Beverages		-	0.23–14.1		
Fast foods and frozen entrees		-	1.86–9.97		
Miscellaneous		-	0.10–92.3		
Herbs and spices, including salt		-	41.5–392		
Canada: Total Diet Studies	2008–2012			Québec City, Calgary, St. Johns, Ottawa, Vancouver data	HC 2019*b*
Dairy products		-	< 0.11–18		
Meat and meat products		-	0.69–16		
Poultry and poultry products		-	0.51–8.9		
Fish and fish products		-	0.48–16		
Soups		-	< 0.02–12		
Cereal and cereal products		-	0.53–16		
Vegetable and vegetable products		-	0.26–17		
Fruit and fruit products		-	0.19–30		
Fats and oils		-	< 0.14–6.7		
Beverages		-	0.25–16		
Fast foods and frozen entrees		-	1.8–18		
Miscellaneous		-	0.26–103		
Herbs and spices, including salt		-	208–470		

Food type	Year	Mean concentration	Range (g/L – water; µg/kg – food)	Comments	Reference
Canada: Total Diet Studies	2016			Québec City data	HC 2019*b*
Dairy products		-	0.12–3.22		
Meat and meat products		-	0.88–11.26		
Poultry and poultry products		-	0.93–3.26		
Fish and fish products		-	0.91–8.73		
Soups		-	1.54–14.83		
Cereal and cereal products		-	0.39–8.27		
Vegetable and vegetable products		-	0.17–14.41		
Fruit and fruit products		-	0.14–22.12		
Fats and oils		-	0.65–4.22		
Beverages		-	< 0.2–6.82		
Fast foods and frozen entrees		-	1.9–4.33		
Miscellaneous		-	< 0.62–65.08		
Herbs and spices, including salt		-	639.27		
Canada: Total Diet Studies	2017-18			Montreal, Calgary data	Government of Canada 2020*b*
Dairy products		-	0.11–8.58		
Meat and meat products		-	0.85-19.61		
Fish and fish products		-	0.60-8.37		
Soups		-	0.79-3.70		
Cereal and cereal products		-	0.47-8.34		
Vegetable and vegetable products		-	0.43-19.03		
Fruit and fruit products		-	ND-14.13		
Fats and oils		-	ND-3.20		
Beverages		-	ND–7.03		
Fast foods and frozen entrees		-	1.14-7.78		
Herbs and spices, including salt		-	57.62-310.03		
Baby food			0.62-5.59		

Food type	Year	Mean concentration	Range (g/L – water; µg/kg – food)	Comments	Reference
Baby food		-	40–140	CFIA Children's Food Chemical's Residue Project	HC 2013*a*
Baby food	2008–2009	-	2–977	CFIA Children's Food Chemical's Residue Project	HC 2013*a*
Infant formula; soy-based	2003–2007	-	2.5–9.1	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2010*a*
Infant formula; milk-based		-	0.5–5.0		
Infant formula; soy-based	2008–2012	-	0.28–4.3	Québec City, Calgary, St. Johns, Ottawa, Vancouver data	HC 2019*b*
Infant formula; milk-based		-	0.04-10		
Infant formula; soy-based	2016	1.36 µg/kg			HC 2019*b*
Infant formula; milk-based		2.09 µg/kg			

Traditional food/country food sampled as part of the First Nations, food, nutrition and environment studies

Food type	Year	Mean concentration	Range (µg/kg)	Comments	Reference
Atlantic Canada					U Ottawa, UdM, AFN 2017
Fish – various species			ND–0.05 µg/g	Range of means. Fresh weight. 21 species. n = 1–11 per species	
Seafood – various species			0.001–0.3 µg/g	Range of means. Fresh weight. n = 1–9 per species	
Harp seal			0.01	Mean. n = 1	
Beaver		ND		µg/g fresh weight. n = 1	
Black bear fat and meat		0.01		µg/g fresh weight. n = 2–3	
Deer meat, liver, heart			0.01–1.4	Range of means. µg/g fresh weight. n = 1–9. Max = 12.2 µg/g for deer meat	
Moose meat and organs			0.01–0.2	Range of means. µg/g fresh weight. n = 1–10.	
Muskrat		0.1		Mean. n = 1	
Rabbit meat and liver			0.1–5.2	Range of means. µg/g fresh weight. n = 1–8.	

Food type	Year	Mean concentration	Range (µg/kg)	Comments	Reference
Squirrel meat			45.4	Mean. n = 2	
Canada goose		0.4		Mean. n = 1	
Grouse		0.2		Mean. n = 11. Max = 1.1	
Berries – various species			ND–0.02	Range of means. µg/g fresh weight. n = 1–11 per species	
Greens/roots/tree foods			ND–3.8	Range of means. µg/g fresh weight. n = 1–8 per species Max = 3.8 in 1 sample of dandelion root	
Garden plants, various root, fruit, stems and seeds			ND–0.001	Range of means. µg/g fresh weight. n = 1–4 per species	
Québec	2013			From animals killed with Pb shot. Concentrations in animals killed by bow or copper shot had much lower concentratoins	Fachehoun et al. 2015
White-tailed deer		0.283 mg/kg	max=4.2	n=35; Pb detected in 90% of samples	
Moose		0.170 mg/kg	max=2.0	n=37; Pb detected in 70% of samples	
Ontario					U Ottawa, UdM, AFN 2014
Fish – various species			ND–0.132	Range of means µg/g. Fresh weight. 21 species. n = 1–18 per species	
Beaver meat, liver			0.044–5.412	µg/g fresh weight. n = 1–10	
Beef		0.004		Mean. n = 1	
Caribou bone, meat			0.007–0.015	Range of means. Fresh weight. n = 1–6	
Deer meat, liver, heart, kidney, tongue			ND–4.905	Range of means. µg/g fresh weight. n = 1–9. Max = 42.4 µg/g for deer meat	
Elk meat		0.011		Mean. n = 1	
Moose meat, marrow, fat and organs			ND–0.985	Range of means. µg/g fresh weight. n = 1–15	
Muskrat meat		0.011		Mean. n = 3	
Rabbit meat, heart, liver			0.006–0.058	Range of means. µg/g fresh weight. n = 1–11	

Food type	Year	Mean concentration	Range (µg/kg)	Comments	Reference
Squirrel meat		1.47		Mean. n = 1	
Red Squirrel meat		0.591		Mean. n = 1	
Waterfowl – various species			0.003–1.562	Range of means. µg/g fresh weight. n = 1–8 per species	
Game birds, terrestrial – various species			0.005–1.204	Range of means. µg/g fresh weight. n = 1–13 per species	
Berries – various species			ND–0.042	Range of means. µg/g fresh weight. n = 1–4 per species	
Mushrooms – various species			0.089–1.19	Range of means. µg/g fresh weight. n = 1 per species	
Leaves – various species			ND–1.100	Range of means. µg/g fresh weight. n = 1–3 per species	
Roots – various species			0.0013–1.07	Range of means. µg/g fresh weight. n = 1–2 per species	
Nuts and seeds – various species			0.005–0.028	Range of means. µg/g fresh weight. n = 1–2 per species	
Stems – various species			ND–0.058	Range of means. µg/g fresh weight. n = 1	
Maple syrup		0.0372		Mean. n = 6	
Manitoba					UNBC, UdM, AFN 2012
Fish – various species			ND–0.140	Range of means µg/g. Fresh weight. 10 species. n = 1–18 per species.	
Beaver meat, liver		0.010		Mean. n = 3	
Caribou meat, organs, gut			ND–0.570	Range of means. Fresh weight. n = 1–2	
Deer meat, liver, heart, kidney			0.025–6.114	Range of means. µg/g fresh weight. n = 1–7. Max = 27.2 µg/g for deer meat	
Elk meat			ND–2.103	Range of means. µg/g fresh weight. n = 1–3. Max = 6.27 µg/g for elk meat	
Moose meat, marrow, fat and organs			0.030–1.619	Range of means. µg/g fresh weight. n = 1–10. Max = 15.6 µg/g for moose meat	
Muskrat meat		ND		Mean. n = 3	

Food type	Year	Mean concentration	Range (µg/kg)	Comments	Reference
Rabbit meat, brain, liver			ND–23.33	Range of means. µg/g fresh weight. n = 1–7. Max = 163 for rabbit meat.	
Waterfowl – various species			ND–1,233.33	Range of means. µg/g fresh weight. n = 1–4 per species Max = 3,700 µg/g in Mallard gizzard.	
Game birds, terrestrial – various species			0.020–152	Range of means. µg/g fresh weight. n = 3–5 per species	
Berries – various species			ND–0.040	Range of means. µg/g fresh weight. n = 1–5 per species	
Leaves, needles, bark – various species			ND–0.250	Range of means. µg/g fresh weight. n = 1–2 per species	
Roots		0.2		Mean. µg/g fresh weight. n = 1	
Alberta					U Ottawa, UdM, AFN 2016
Fish – various species			ND–0.020	Range of means µg/g. Fresh weight. 9 species. n = 1–4 per species	
Beaver meat		0.019		Mean. n = 1	
Bison meat, liver, kidney			0.009–32.75	Range of means. µg/g fresh weight. n = 1–2. Max = 65.5 µg/g for bison meat	
Black bear meat		0.013		Mean. n = 1	
Deer meat, liver, fat, kidney			0.004–0.099	Range of means. µg/g fresh weight. n = 1–7	
Elk meat		0.078		Mean. µg/g fresh weight. n = 5	
Moose meat, gut, tongue and organs			0.004–0.098	Range of means. µg/g fresh weight. n = 1–8	
Porcupine meat		0.115		Mean. n = 1	
Rabbit meat, liver			0.008–4.2	Range of means. µg/g fresh weight. n = 1–7. Max = 27.3 for rabbit meat	
Waterfowl – various species			ND–0.486	Range of means. µg/g fresh weight. n = 1–6 per species	
Game birds, terrestrial – various species			0.963–1.198	Range of means. µg/g fresh weight. n = 3–10 per species	
Berries – various species			ND–0.018	Range of means. µg/g fresh weight. n = 1–6 per species	

Food type	Year	Mean concentration	Range (µg/kg)	Comments	Reference
Leaves, needles, bark – various species			ND–0.386	Range of means. µg/g fresh weight. n = 1–4 per species	
Roots – various species			ND–0.002	Mean. µg/g fresh weight. n = 1–4	
British Columbia					UNBC, UdM, AFN 2011
Fish – various species			ND–0.15	Range of means µg/g. Fresh weight. 24 species. n = 1–12 per species	
Seafood – various species			ND–0.19	Range of means µg/g. Fresh weight. 15 species. n = 1–3 per species	
Bear meat, fat, liver			0.06–0.73	Range of means. µg/g fresh weight. n = 1–2	
Beaver meat, fat, organs			ND–2.69	Range of means. µg/g fresh weight. n = 1–4	
Bison meat, liver, kidney		ND		Mean. µg/g fresh weight. n = 1	
Buffalo meat		0.24		Mean. µg/g fresh weight. n = 1	
Deer meat, liver, heart			ND–1.49	Range of means. µg/g fresh weight. n = 1–15. Max = 13.9 in deer meat	
Elk meat, fat kidney, liver			ND–0.03	Range of means. µg/g fresh weight. n = 1–6	
Goat meat		0.13		Mean. n = 2	
Groundhog meat		0.06		Mean. n = 1	
Moose meat, marrow, fat, gut, tongue and organs			ND–0.17	Range of means. µg/g fresh weight. n = 1–15	
Rabbit meat		0.24		Mean. n = 6	
Waterfowl – various species			ND–2.65	Range of means. µg/g fresh weight. n = 1 per species	
Game birds - grouse		13.15		Range of means. µg/g fresh weight. n = 8. Max = 60.6 µg/g	
Berries – various species			ND–0.04	Range of means. µg/g fresh weight. 25 species. n = 1–11 per species	
Leaves, needles – various species			ND–1.90	Range of means. µg/g fresh weight. 16 species, n = 1–7 per species	

Food type	Year	Mean concentration	Range (µg/kg)	Comments	Reference
Roots – various species			ND	Range of means. µg/g fresh weight. n = 1–2 per species	
Stem – various species			ND–0.07	Range of means. µg/g fresh weight. n = 1–4 per species	
Bark – various species			ND–0.09	Range of means. µg/g fresh weight. n = 1 per species	
Sap – various species			ND–0.07	Range of means. µg/g fresh weight. n = 1 per species	
Nuts			ND	Mean. n = 1.	
Mushrooms – various species			ND–0.08	Range of means. µg/g fresh weight. n = 1–3 per species	
Seaweed			ND–0.48	Range of means. µg/g fresh weight. n = 1–3 per species	

Human milk for breast-fed infants

Location	n	Mean concentration (SD) µg/L	Range	Comments	Reference
Canada	210	1.04	0.025–15.8	Assumed mature milk (not specified)	Dabeka *et al*. 1986
Canada: NL	17	Median: 0 (0)		Colostrum, full-term gestation	Friel *et al*. 1999
	119	Median: 2.43 (13.63)		Mature milk, full-term gestation	
	24	Median: 0 (0)		Colostrum, preterm gestation	
	12–21	Median: 0–1 (0–1.48)		Mature milk, preterm gestation	
Canada: Northern ON	25	2.08 (1.67)	0.41–8.33	Mushkegowuk territory First Nations community	Hanning *et al*. 2003
United States: Boston, MA	100	17 (20)	0–72	1 and 6 months postpartum	Rabinowitz *et al*. 1985
NJ	135	6.1 (11.62)	-	1.5 months postpartum	Sowers *et al*. 2002
		5.6 (12.78)		3 months postpartum	
		5.9 (11.62)		6 months postpartum	
		4.3 (18.59)		12 months postpartum	
Sweden	35	0.5 (0.3)	-	Control Area (relative to exposure smelter area)	Palminger Hallén *et al*. 1995

Location	n	Mean concentration (SD) µg/L	Range	Comments	Reference
Austria: Tulln	48	1.22 (0.92)	-		Gundacker *et al*. 2002
Linz	42	2.48 (2.39)	-		
Vienna	45	1.29 (1.12)	-		
Overall	138	1.63 (1.66)	-		
Portugal	34	20.1 (1.38)	0.06–5.43	Colostrum	Almeida *et al*. 2008
	19	0.94 (1.05)	0.07–4.03	Mature milk	
Australia: Sydney	48	0.73 (0.7)	0.9–3.1	Mature milk	Gulson *et al*. 1998

Human fluids

Location	Tissue	Year	Geometric mean concentration (µg/dL)	Range	n	Comments	Reference
Canada	Urine	2007–09	0.48	0.13–2.11	5,492	3–79 yrs	HC 2010*d*
	Urine	2009–11	0.52		6,311	3–79 yrs	HC 2013*c*
	Whole blood	2007–09	1.34	0.60–3.79	5,319	6–79 yrs. Range = 10^{th}-95^{th} percentiles.	HC 2010*d*
	Whole blood	2009–11	1.2	0.54–3.2	6,070	3–79 yrs. Range = 10^{th}-95^{th} percentiles.	HC 2017*A*
	Whole blood	2012–13	1.1	0.49–3.2	5,538	3–79 yrs. Range = 10^{th}-95^{th} percentiles.	
	Whole blood	2014–15	0.95	0.43–2.7	5,498	3–79 yrs. Range = 10^{th}-95^{th} percentiles.	
	Whole blood	2016-17	0.93	0.39-2.5	4,517	3–79 yrs. Range = 10^{th}-95^{th} percentiles	HC 2019*c*
Canada, NU	Whole blood	2012	71.1	3.9-898.9	100	≥20 yrs. 35 individuals >100 µg/dL. Subjects identified from households where at least one individual had preveiously presented >100 µg/dL.	Fillion et al. 2014
	Whole blood	2012	17.5	6.0-440.9	56	≤10 yrs. 1 individual >100 µg/dL. Subjects identified from households where at least one individual had preveiously presented >100 µg/dL.	

Consumer products

Location	Product	Year	Mean concentration (mg/kg)	Range	Comments	Reference
Canada	Dietary supplements	2004–05	0.21–1.6	0.17–8.4	Marine origin supplements available in Canada	Leblond *et al*. 2008
United States	Vitamins	-	0.062 (0.075)	ND–0.395	0–6-year-old children	U.S. FDA 2008
		-	0.195 (0.149)	0.005–0.623	≥ 7-year-old children	
		-	0.258 (0.185)	0.007–1.20	Adult women	
		-	0.291 (0.304)	ND–2.40	Pregnant and lactating women	
New Orleans, LA	Paint	-	Median 35.2	0.112–256.8		Mielke *et al*. 2001
Unknown	Coal	-	11 (37)	1,900 (max)		Finkelman 1999

APPENDIX 2. BIOAVAILABILITY OF LEAD IN SOILS OF VARYING MINERALOGY

Soil Group	Estimated Relative Bioavailability	Reference
Pb(M) oxide	0%	Casteel *et al*. 2006
Fe(M) sulphate	0%	
Galena	3.3%	
Anglesite	5.3%	
Fe(M) oxide	24.1%	
Pb phosphate	46.6%	
PbO	51.6%	
Mn(M) oxide	94.2%	
Cerussite	99.2%	
US EPA IEUBK Default	60%	