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 ZINC

Protection of Environmental and Human Health

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NOTE TO READERS

The Canadian Council of Ministers of the Environment (CCME) is the primary minister-led intergovernmental forum for collective action on environmental issues of national and international concern.

This document provides the background information and rationale for the development of Canadian Environmental Soil Quality Guidelines for zinc. For additional technical information regarding these guidelines, please contact:

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These guidelines are included as updates in the *Canadian Environmental Quality Guidelines*, which was published by the CCME in October of 1999. The Canadian Environmnental Quality Guidelines are available online at http://ceqg-rcqe.ccme.ca.

This scientific supporting document is available in English only. Ce document scientifique du soutien n'est disponible qu'en anglais avec un résumé en français. Un sommaire de cette information technique est disponible en français sous le titre *Recommandations canadiennes pour la qualité de l'environnement* (CCME 1999; http://ceqg-rcqe.ccme.ca/).

Reference listing:

Canadian Council of Ministers of the Environment (CCME). 2018. Scientific Criteria Document for the Development of the Canadian Soil Quality Guidelines for Zinc: Protection of Environmental and Human Health. CCME, Winnipeg, MB.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF APPENDICES

LIST OF FREQUENTLY USED ACRONYMS AND ABBREVIATIONS

AAS	Atomic Absorption Spectrophotometry
ACGIH	American Conference of Industrial Hygienists
ATSDR	Agency for Toxic Substances and Disease Registry
BCF	bioconcentration factor
BF	bioavailability factor
BSC	background soil concentration
BW	body weight
CEC	cation exchange capacity
CCME	Canadian Council of Ministers of the Environment
CMBEEP	Committee on Medical and Biological Effects of Environmental Pollutants
DRI	dietary reference intakes
DTED	daily threshold effect dose
EC	Environment Canada
EC_{xx}	effects concentration
ECCC	Environment and Climate Change Canada
ECL	effects concentration—low
EDI	estimated daily intake
EFSA	European Food Safety Authority
ESOD	**Erythrocyte** superoxide dismutase
ESSD	estimated species sensitivity distribution
ETE	essential trace elements
FIR	food ingestion rate
DL	detection limit
DMIR	dry matter intake rate
DWSP	Drinking Water Surveillance Program
GFAA	Graphite Furnace Atomic Absorption
GSC	Geological Survey of Canada
HC	Health Canada
HDL	high-density lipoproteins
IC	inhibition concentration
ICP-AES	inductively coupled plasma atomic emission spectroscopy
ICP-MS	inductively coupled plasma mass spectrometry
INAA	Instrumental Neutron Activation Analysis
I/O	indoor/outdoor
IOM	Institute of Medicine
JECFA	Joint Expert Committee on Food Additives
LC50	lethal concentration (resulting in the death of 50% of the population)
LDL	low density lipoprotein
LOAEL	lowest observed adverse effect level
LOEC	lowest observed effects concentration
MRL	minimal risk level
NAPS	National Air Pollution Surveillance
NASGLP	North American Soil Geochemical Landscape Project
NECC	nutrient and energy cycling check

NHEXAS	National Human Exposure Assessment Survey
NOEC	no observed effect concentration
NPRI	National Pollutant Release Inventory
NRC	National Research Council
NRCan	Natural Resources Canada
OC	organic carbon
OM	organic matter
OMEE	Ontario Ministry of the Environment and Energy
OMOECC	Ontario Ministry of the Environment and Climate Change
OSHA	Occupational Safety and Health Administration
OTR	Ontario Typical Range
PDF	probability density functions
PGMN	Provincial Groundwater Monitoring Network
PM 25	particulate matter less than 2.5 μm in diameter
PPM	parts per million
PSI	proportion of soil ingested
RAF	relative absorption factors
RfD	reference dose
RTDI	residual tolerable daily intake
RIVM	National Institute of Public Health and the Environment (the Netherlands)
SAF	soil allocation factor
SIR	soil ingestion rate
SQG	soil quality guideline
SQG_{DH}	soil quality guideline for direct human health
SQG_{E}	soil quality guideline for environmental health
SQG_{HH}	soil quality guideline for human health
SQG_{I}	soil quality guideline for food ingestion
SQG_{NEC}	soil quality guideline for nutrient and energy cycling
SQG_{OM-E}	soil quality guideline for off-site migration
SQG_{SC}	soil quality guideline for soil contact
TEC	threshold effects concentration
TDI	tolerable daily intakes
TDS	Total Diet Study
TRV	toxicity reference value
UF	uncertainty factor
UL	upper intake levels
US EPA	United States Environmental Protection Agency
WHO	World Health Organization

EXECUTIVE SUMMARY

Canadian environmental quality guidelines 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 can be used as the basis for consistent assessment and remediation of contaminated sites in Canada.

These guidelines were developed according to the procedures described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 2006). According to this protocol, both environmental and human health soil quality guidelines are developed and the lowest value generated of the two approaches for each of the four land uses is recommended by CCME as the Canadian Soil Quality Guidelines (CCME 2006).

This scientific criteria document provides the background information and rationale for the derivation of environmental and human health soil quality guidelines for zinc. This document contains a review of information on the chemical and physical properties of zinc; the sources and emissions in Canada; the distribution and behaviour of zinc in the environment; the metabolic fate, behaviour and toxicological effects of zinc on microbial processes, plants, invertebrates, livestock and wildlife; and the behaviour and effects in humans and mammalian species. This information is used to derive soil quality guidelines for zinc to protect ecological and human receptors in four types of land uses: agricultural, residential/parkland, commercial and industrial.

The environmental soil quality guidelines for zinc for each of the four land uses are: 250 mg/kg soil for agricultural land use, 250 mg/kg soil for residential/parkland land use, 410 mg/kg soil for commercial land use and 410 mg/kg soil for industrial land use. These guidelines are protective of ecological receptors and are optimized for soils within the pH range of 4 to 8.3, because the toxicological studies upon which they are based were conducted within this pH range. The environmental soil quality guidelines were selected from the following ecological exposure pathways for zinc: soil contact, soil and food ingestion, nutrient and energy cycling check, and off-site migration check.

The human health-based soil quality guidelines for zinc for each of the four land uses are: 10,000 mg/kg soil for agricultural land use, 10,000 mg/kg soil for residential/parkland land use, 16,000 mg/kg soil for commercial land use and 140,000 mg/kg soil for industrial land use. The human-health-based soil quality guidelines were selected from direct human-health-based soil quality guidelines for soil ingestion, particulate inhalation and dermal contact and the off-site migration check.

The Canadian Soil Quality Guidelines for the protection of environmental and human health, as recommended by the Canadian Council of Ministers of the Environment (CCME 2006) are based on the lowest of the environmental soil quality guidelines or the human-health-based soil quality guidelines. This revision to the Canadian Soil Quality guidelines for zinc supersedes the Canadian zinc soil quality guideline derived in 1999 (Environment Canada [EC] 1999a), first published in CCME (1999), and the zinc Interim remediation criteria for soil (CCME 1991).

RÉSUMÉ

Les recommandations canadiennes pour la qualité de l'environnement sont des concentrations exprimées en valeurs numériques ou des énoncés circonstanciés visant à assurer un écosystème sain, capable de soutenir les utilisations actuelles et probables du site par les récepteurs écologiques et humains. Les recommandations canadiennes pour la qualité des sols peuvent être utilisées comme base pour une évaluation et un assainissement conformes des lieux contaminés au Canada.

Ces recommandations ont été élaborées selon les procédures décrites dans le *Protocole d'élaboration de recommandations pour la qualité des sols en fonction de l'environnement et de la santé humaine* (CCME 2006). Les recommandations pour la qualité des sols visant la protection de l'environnement et de la santé humaine sont élaborées conformément à ce protocole, et la plus petite valeur obtenue dans le cadre des deux approches pour chacune des quatre vocations de terrain est recommandée par le CCME comme étant la recommandation canadienne pour la qualité des sols (CCME 2006).

Le présent document scientifique contient des renseignements généraux et la justification de la détermination des recommandations pour la qualité des sols visant la protection de l'environnement et de la santé humaine contre le zinc. Le document contient une revue de l'information sur les points suivants : les propriétés chimiques et physiques du zinc; les sources et les émissions au Canada; la distribution et le comportement du zinc dans l'environnement; son devenir métabolique, son comportement et ses effets toxicologiques sur les procédés microbiens, les plantes, les invertébrés, les animaux d'élevage et la faune; ainsi que son comportement et ses effets chez les humains et les espèces mammifères. Cette information sert à l'élaboration des recommandations pour la qualité des sols concernant le zinc afin de protéger les récepteurs écologiques et humains dans quatre types de vocation de terrain : agricole, résidentielle/parc, commerciale et industrielle.

Les recommandations pour la qualité des sols concernant le zinc visant la protection de l'environnement pour chacune des quatre vocations de terrain sont les suivantes : 250 mg/kg de sol pour les terrains à vocation agricole, 250 mg/kg de sol pour les terrains à vocation résidentielle/parc, 410 mg/kg de sol pour les terrains à vocation commerciale et 410 mg/kg de sol pour les terrains à vocation industrielle. Ces recommandations sont protectrices des récepteurs écologiques et optimisées pour les sols ayant un pH allant de 4 à 8,3, puisque les études toxicologiques utilisées pour leur élaboration ont été effectuées dans cette plage de pH. Les recommandations pour la qualité des sols concernant le zinc visant la protection de l'environnement ont été sélectionnées parmi les voies d'exposition écologiques suivantes pour le zinc : le contact avec le sol, l'ingestion de sol et de nourriture, la vérification portant sur les cycles des nutriments et de l'énergie et la vérification portant sur la migration hors site.

Les recommandations pour la qualité des sols concernant le zinc visant la protection de la santé humaine pour chacune des quatre vocations de terrain sont les suivantes : 10 000 mg/kg de sol pour les terrains à vocation agricole, 10 000 mg/kg de sol pour les terrains à vocation résidentielle/parc, 16 000 mg/kg de sol pour les terrains à vocation commerciale et 140 000 mg/kg de sol pour les terrains à vocation industrielle. Les recommandations pour la qualité des sols visant la protection de la santé humaine ont été sélectionnées parmi les recommandations pour la qualité des sols relative au contact direct (santé humaine) pour l'ingestion de sol, l'inhalation de particules et le contact cutané, et la vérification portant sur la migration hors site.

Les recommandations canadiennes pour la qualité des sols visant la protection de l'environnement et de la santé humaine, telles que recommandées par le Conseil canadien des ministres de l'environnement (CCME 2006), sont basées sur les recommandations pour la qualité des sols visant la protection de l'environnement ou de la santé humaine ayant les niveaux les plus bas. Les présentes recommandations canadiennes révisées pour la qualité des sols concernant le zinc remplacent la recommandation canadienne pour la qualité des sols élaborée en 1999 (Environnement Canada [EC] 1999a) et initialement publiée dans CCME (1999), ainsi que les critères provisoires d'assainissement du sol concernant le zinc (CCME 1991).

1.0 INTRODUCTION

Canadian Environmental Quality Guidelines are intended to protect, sustain, and enhance the quality of the Canadian environment and its many beneficial uses. They are generic numerical concentrations or narrative statements that specify levels of toxic substances or other parameters in the ambient environment that are recommended to protect and maintain wildlife and/or the specified uses of water, sediment and soil. These values are nationally endorsed through the Canadian Council of Ministers of the Environment (CCME) and are recommended for toxic substances and other parameters (e.g., nutrients, pH) of concern in the ambient environment.

The development of the Canadian Soil Quality Guidelines was initiated by CCME in 1991. In response to the urgent need to begin remediation of high priority "orphan" contaminated sites, an interim set of soil quality criteria was adopted from values that were in use in various jurisdictions across Canada (CCME 1991). There is a continued need for national soil quality guidelines for the management of soil quality with a particular focus on remediation of contaminated sites. The Canadian Soil Quality Guidelines are developed according to procedures that have been described by CCME (CCME 1996a; revised in 2006). According to this 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 Canadian Soil Quality Guideline. The original Canadian Soil Quality Guidelines for zinc were released by CCME in a working document entitled *Recommended Canadian Soil Quality Guidelines* (CCME 1997), which was then revised and published by CCME in 1999 (CCME 1999); its derivation was documented in a supporting document developed by Environment and Climate Change Canada (formerly Environment Canada) (EC 1999a). The current revision (this document) to the Canadian Soil Quality Guidelines for zinc supersedes the 1999 and 1997 zinc soil quality guidelines and the 1991 interim remediation criteria for soil (CCME 1991). The interim remediation criteria for soil (CCME 1991) should be used only when soil quality guidelines based on the CCME protocol (CCME 1996a or 2006 update) have not yet been developed for a given chemical.

This scientific supporting document provides the background information and rationale for the derivation of environmental soil quality guidelines for zinc. This document contains a review of information on the chemical and physical properties of zinc; sources and emissions in Canada; the distribution and behaviour of zinc in the environment; the toxicological effects of zinc on microbial processes, plants, invertebrates, livestock and wildlife; and the behaviour and effects in humans and mammalian species. This information is used to derive soil quality guidelines for zinc to protect human and ecological receptors in four types of land uses: agricultural, residential/parkland, commercial and industrial (CCME 2006). The current revision to the environmental soil quality guidelines builds upon toxicity data first reported in the original zinc scientific supporting document (EC 1999a), whereas the derivation of human health soil quality guidelines for zinc represents new work that first appears in this document.

The Canadian Soil Quality Guidelines presented in this document are intended as general guidance. Site-specific conditions should be considered in the application of these values (see CCME 1996b for guidance on developing site-specific soil objectives). The reader is referred to

CCME (2006) for further generic implementation guidance pertaining to the guidelines. Soil quality guidelines are derived to approximate a “no- to low-” effect level (or threshold level) based only on the toxicological information and other scientific data (fate, behaviour, etc.) available for the substance of concern. They do not consider socioeconomic, technological and political factors or lifestyle choices. These non-scientific factors are to be considered by site managers at the site-specific level as part of the risk management process. Because these guidelines may be used and applied differently across provincial and territorial jurisdictions, the reader should consult the laws and regulations of the jurisdiction they are working within for applicable implementation procedures.

2.0 BACKGROUND INFORMATION

2.1 Physical and Chemical Properties

Zinc (Zn, CAS # 7440-66-6) is a transition metal, which is a ductile and malleable element that conducts electricity and heat. Bluish-white zinc is not found as free metal in the natural environment, but rather occurs as a mineral in the earth crust (Natural Resources Canada [NRCan] 2006).

Zinc tends to strongly react with organic and inorganic compounds. Zinc can also form stable combinations with many organic substances such as humic and fulvic acids and a wide range of biochemical compounds. At American hazardous sites, zinc has been found as zinc chloride, zinc oxide (zincite), zinc sulphate and zinc sulphide (sphalerite and wurtzite) (Agency for Toxic Substances and Disease Registry [ATSDR] 1995).

Metallic zinc is insoluble, while the solubilities of different zinc compounds range from insoluble (oxides, carbonates, phosphates and silicates) to extremely soluble (sulphates and chlorides).

Zinc has five stable and six radioactive isotopes. This document only addresses the stable isotopes. Some relevant physical and chemical properties of zinc and compounds are presented in Table 1.

Table 1: Physical and Chemical Properties of Zinc and its Common Salts

Properties (Units)	Zinc	Zinc Oxide	Zinc Sulphide (wurtzite)	Zinc Sulphate	Zinc Chloride	Zinc Fluoride	Zinc Bromide	Zinc Iodide	Zinc Acetate dihydrate	Zinc Borate	Zinc Carbonate	Zinc Chromate
Chemical Formula	Zn	ZnO	ZnS	$ZnSO_4$	$ZnCl_2$	ZnF_2	$ZnBr_2$	ZnI_2	$Zn(C_2H_3O_2)_2 \cdot 2H_2O$	$3ZnO \cdot 2B_2O_3$	$ZnCO_3$	$ZnCrO_4$
CAS no.	7440-66-6	1314-13-2	1314-98-3	7733-02-0	7646-85-7	7783-49-5	7699-45-8	10139-47-6	5970-45-6	27043-84-1	3486-35-9	13530-65-9
Molecular Weight ($g \cdot mol^{-1}$)	65.38	81.41	97.47	161.47	136.32	103.41	225.22	319.22	219.53	383.47	125.42	181.4
Physical state	Blue-white, metal; hexagonal	White powder; hexagonal	White hexagonal crystals	Colourless orthorhombic crystals	White hygroscopic crystals	White tetragonal needles, hygroscopic	White hexagonal or hygroscopic crystals	White-yellow hygroscopic crystals	White powder	White amorphous powder	White hexagonal crystals	Yellow prisms
Boiling point (°C)	907	ND	sublimes	ND	732	1,500	≈670	625	ND	ND	ND	ND
Melting point (°C)	419.53	1,974	1,700	680 decomposes	290	872	402	450	237 decomposes	ND	140 decomposes	316
Density (g/cm^3)	7.13	5.6	4.09	3.8	2.91	4.90	4.50	4.74	1.74	3.64	4.43	3.40
Solubility (g/100 g H_2O)	ND	ND	ND	57.7	408	1.55	488	438	30 (20°C)	ND	0.000091 (20°C)	3.08

Source: Haynes 2015
Solubility is at 25°C unless otherwise specified
ND = no data

2.2 Geochemical Properties

Zinc is a trace element, or a trace metal, that occurs in natural environments and also from anthropogenic inputs. It is one of the most common elements (25th) in the Earth's crust (approximately 0.004 per cent). The worldwide average concentration in the Earth's crust and soils is estimated to be 70 mg/kg (Kabata-Pendias 2011). It can be found in air, soil and water and is present in foods. Many rocks and minerals contain zinc in varying amounts. Its abundance in different minerals is influenced by the zinc concentration of the magma and premetamorphic rock as well as the ability of the crystal lattice to incorporate this element. Zinc is distributed rather uniformly in magmatic rocks, it increases slightly in mafic rock and its distribution is decreased in acid rock (Kabata-Pendias and Pendias 2000), whereas it concentrates in argillaceous sedimentary rock (up to 120 mg/kg) (Kabata-Pendias 2011). The average worldwide concentration in igneous rocks is 65 ppm (1 ppm=1 mg/kg) with a range of 5–1,070 ppm. For limestone, the average concentration is 20 ppm with a range of <1–180 ppm. For sandstone, the average concentration is 30 ppm with a range of 5–170 ppm.

Most zinc is distributed as a minor constituent of rock-forming minerals. Zinc ore is frequently found in the environment in the form of zinc sulphide (ZnS) and sphalerite (zinc iron sulphide) minerals. The most common impurities in zinc minerals are Fe, Cd and Pb (Adriano 2001).

2.3 Analytical Methods

Analytical guidance for Canadian environmental quality guidelines can be found in *Guidance Manual for Environmental Site Characterization in Support of Environmental and Human Health Risk Assessment Volume 4: Compendium of Analytical Methods for Contaminated Sites* (CCME 2016). Volume 4 provides advice on sample preparation and analysis for zinc in soil, and recommended laboratory reporting limits.

ICP-AES and ICP-MS[1] are the methods most widely used used on environmental samples. Several of these analytical methods are adapted to various environmental media, including solid, liquid and sludge samples. The most recent version of the ICP-AES method approved by the United States Environmental Protection Agency (US EPA) for analysis of metals is method 6010C (US EPA 2007b); for the ICP-MS method 6020B is the most recent (US EPA 2014), although US EPA method 200.8 (US EPA 1994) is also still commonly used; methods 7000B and 7010 are the most recent methods for Atomic Absorption Spectrophotometry (AAS) and Graphite Furnace Atomic Absorption (GFAA), respectively (US EPA 2007c; 2007d). These methods can be used for soil, water and sediment samples, although testing of soil samples also involves an initial acid digestion, generally based on US EPA method 3050B (US EPA 1996),

[1] ICP-AES: inductively coupled plasma atomic emission spectroscopy; ICP-MS: inductively coupled plasma mass spectrometry

which is the method used by the Geological Survey of Canada (GSC) for the digestion of samples for zinc determination in soil and sediment. Detection limits of below parts per billion levels can be achieved using ICP-MS (Feng 2008); however, practical detection limits and precisions (by relative standard deviation) for environmental samples commonly range from 0.1 to 10 mg/kg at 14–42 per cent for soil and 0.001 to 0.5 mg/L at 6.8–17 per cent for water samples (US EPA 2014). Other instrumental methods previously published by the US EPA and in the *Standard Methods for the Examination of Water and Wastewater* are also currently in use. In general, soil concentrations derived from aqua regia extraction are more relevant from a human health perspective.

The estimated instrumental detection limit (DL) of US EPA method 6010C is 15 µg/L (1 g of soil digested in 100 mL water). US EPA analytical method 6020, *Inductively Coupled Plasma with Mass Spectrometry,* is a more sensitive technique, with an estimated instrumental detection limit (DL) of <0.02 µg/L (US EPA 2014). This method is applicable to groundwater, aqueous samples, industrial wastes, soils, sludges, sediments and other solid wastes.

The quality of the most recent data (2003 to 2009) from the National Air Pollution Surveillance (NAPS) database has improved with better method detection limits achieved with the use of ICP-MS analyses (Dann 2010).

2.4 Production and Uses in Canada

Most of the zinc produced globally comes from ores containing zinc sulphide minerals. There are more than 80 zinc minerals known, but there are only a few important commercial ores. Sphalerite is the most important ore mineral and the principal source of zinc metal for the zinc industry (Adriano 2001). The total world mine production of zinc in 2014 was 13.2 Mt, while refined zinc production was 13.3Mt (NRCan 2015). For that same period, Canada's mining production was 352,745 tonnes (up 17 per cent from 2013), and refined production was 647,881 tonnes (down 0.6 per cent from 2013) (NRCan 2015). Canada is the fourth largest producer of zinc globally.

In Canada, zinc is produced from mines in Yukon, British Columbia, Manitoba, Ontario, Québec, and Newfoundland and Labrador (NRCan 2015). Canada has a total refining capacity of 693,000 tonnes (NRCan 2014). Zinc metal is produced at three metallurgical sites in Québec, Manitoba and British Columbia. Zinc oxide is produced at one plant in Ontario (NRCan 2014). Canadian development of zinc-producing mines is ongoing in Newfoundland, New Brunswick, Québec, Yukon and Nunavut (NRCan 2014). However, Canadian zinc production is predicted to decrease mainly due to the closure of a large mine in New Brunswick in 2013, projected to result in a loss of production of about 200,000 tonnes annually (NRCan 2014). Despite Canada's decrease in production, zinc demand continues to increase (primariliy for galvanized steel). This demand will be met internationally (NRCan 2014).

The greatest use of zinc is as a protective coating (galvanization) for iron and steel products (50 per cent of worldwide zinc use) to prevent rusting or corrosion, the majority of which is used in the automobile industry. The zinc protective coating is applied electrolytically or by hot dip methods. Zinc is also used for a wide range of die-cast products (17 per cent of worldwide zinc uses), including automobile components, small electrical appliances, business machines, tools and toys. Zinc is an essential component of brass (an alloy of copper and zinc). Zinc brasses have

good physical, electrical and thermal properties and are corrosion resistant, making them useful in plumbing, heat exchange equipment and other hardware (NRCan 2014).

2.5 Sources and Concentrations in the Canadian Environment

The evaluation of levels of zinc in air, soil, water, sediment and biota provides a means of determining the routes and magnitudes of exposures to environmental receptors. The background concentrations and environmental fate of metals strongly depend on geological and biological characteristics, and therefore any assessment of potential risks associated with metals should take into consideration regional differences in metal content in the natural environment (Chapman and Wang 2000).

Worldwide, anthropogenic and natural sources of zinc are estimated to be of similar magnitude, although it is difficult to estimate a ratio of natural to anthropogenic releases due to uncertainties in estimating natural sources. Erosion is the largest natural source of zinc to water (World Health Organization [WHO] 2001). Natural inputs to air mainly result from igneous emissions and forest fires. The main anthropogenic sources of zinc include: mining, zinc production facilities, iron and steel production, corrosion of galvanized structures, coal and fuel combustion, waste disposal and incineration, and the use of zinc-containing fertilizers and pesticides. In general, anthropogenic sources of zinc have decreased since the 1970s due to more efficient emission technologies (WHO 2001).

Additional sources of anthropogenic zinc to the environment include: electroplaters, mine drainage, domestic and industrial sewage, combustion of solid wastes and fossil fuels, road surface runoff, corrosion of zinc alloy and galvanized surfaces, intensive animal production, and erosion of agricultural soils (ATSDR 2005; EC 1999a; Mohanna and Nys 1999). Tire debris may also release significant quantities of zinc into soil (Smolders and Degryse 2002). According to the National Pollutant Release Inventory (NPRI), 1,085 tonnes of zinc and zinc compounds was released to the environment by major industrial sources in 2013, of which 476 tonnes was released to air, 255 tonnes to water and 354 tonnes to land (EC 2015). The major emitters were metallurgical facilities. NPRI only includes data for facilities with 20,000 annual reported employee hours that manufacture, process or use at least 10 tonnes of zinc, and therefore may not represent all anthropogenic emissions in Canada (EC 2013).

Concentrations of zinc in environmental media are summarized in Appendix I.

2.5.1 Ambient Air

Zinc concentrations in air in Canada are summarized in Appendix I.

The majority of zinc emissions to the atmosphere come from mining operations. Other major emitters are zinc metal production, cement production, the use of phosphate fertilizers and, to a lesser degree, the incineration of oil, coal and refuse (Nriagu and Pacyna 1988 in WHO 2001). Modern production techniques have greatly lowered emissions since the 1970s (WHO 2001).

It has been estimated that 4,600 tonnes (95 per cent confidence interval: 1,200–10,000 tonnes) of zinc is released to the atmosphere in Canada by natural processes, with the largest natural source

being wind-blown dust. Emissions from sea salt spray may also be significant, particularly in coastal regions; forest fires can also be a source of natural zinc emissions (Richardson *et al.* 2001).

Data on Canadian zinc concentrations in air in Canada, reported as $PM_{2.5}$ (particulate matter less than 2.5 μm in diameter) were provided by Environment and Climate Change Canada from the NAPS, a Canada-wide network of monitoring stations operated by federal, provincial, territorial and municipal governments and agencies. The data set most relevant to zinc exposure via inhalation is the respirable size fraction (i.e., $PM_{2.5}$) analyzed by ICP-MS following acid (HNO_3) digestion (Health Canada [HC] 2011a). Sulphate, iron and zinc in $PM_{2.5}$ are associated with mortality (zinc having the strongest association) (Burnett *et al.* 2000). Based on the 2003–2009 NAPS dataset, the overall mean concentration of zinc in $PM_{2.5}$ from urban and rural stations was 0.013 μg/m^3 (n=3,053 samples) (HC 2011a). The mean concentration of $PM_{2.5}$ calculated from the 2003–2009 NAPS dataset was used to estimate typical zinc concentrations in ambient air in Canada and to estimate the estimated daily intake (EDI) by the Canadian population.

Additional data sources for zinc concentrations in ambient air can provide more information on regional variations and are provided here for information purposes. The following data were not used to calculate the estimate of typical concentrations in ambient air in Canada.

A survey by Burnett *et al.* (2000) from 1986 to 1996 found the mean zinc concentration associated with $PM_{2.5}$ in eight Canadian cities to be 0.0258 μg/m^3. Slightly higher concentrations were measured, in Windsor, ON by Niu *et al.* (2009): zinc concentrations of 0.050-0.382 μg/m^3 were found in samples collected in summer of 2005 and measured by ICP-MS. Generally, atmospheric zinc concentrations are higher in summer than winter, and are also higher near point sources (such as iron- and steel-producing factories and zinc, lead and copper smelters) (ATSDR 2005).

2.5.2 Indoor Air

Available data on concentrations of zinc in indoor air are summarized in Appendix I.

Limited indoor air data from Canadian sources are available. As such, data from non-Canadian sources (Adgate *et al.* 2007; Balasubramanian and Lee 2007; Graney *et al.* 2004; Li *et al.* 1993; Molnar *et al.* 2006; Stranger *et al.* 2009; Van Winkel and Scheff 2001) were considered, in addition to one Canadian source of indoor air data (Alberta Health 1998). Using this expanded dataset, a mean indoor air concentration of 0.015 ± 0.017 ng/m^3 (mean ± SD) was estimated as representative of indoor air concentrations in Canada (HC 2011a) for the purposes of developing the inhalation EDI for zinc, used in the human health soil quality guideline calculation. The EDI methodology is further dicussed in Section 7.2 of this document.

2.5.3 Indoor Settled Dust

Two Canadian studies examining the concentration of zinc in indoor settled dust were identified (Rasmussen *et al.* 2001; 2008). In addition to these two Canadian studies, data were obtained from the National Human Exposure Assessment Survey (NHEXAS), a program initiated by the US EPA Office of Research and Development (US EPA 2009) and from other developed

countries (Adgate *et al.*1998; Chattopadhyay *et al.* 2003; Davis and Gulson 2005, Kim and Fergusson 1993; Lisiewicz *et al.* 2000; Madany et. al. 1994; Seifert *et al.* 2000; Turner and Simmonds, 2006; Yaghi and Abdul-Wahab 2004;). The study by Seifert *et al.* (2000) was part of the German Environmental Survey, which is a program similar to NHEXAS. Although these U.S. and German programs are not Canadian sources, they are large, well-designed studies, therefore it was advantageous to use the above-mentioned data in order to provide more data. From these studies, the arithmetic mean zinc concentration in Canadian dust is calculated to be 1,278 mg/kg (SD=6,649, n=4,781) (HC 2011a). These data were used to develop the dust exposure portion of the zinc EDI values used in human health SQG calculations (Section 7.2).

There are many factors that can greatly influence settled dust concentrations, such as cleaning frequency, presence of pets, whether shoes are worn indoors, etc. that make it difficult to compare data. From the two Canadian studies (Rasmussen *et al.* 2001; 2008), the average zinc concentrations in indoor dust ranged from 717 mg/kg (2001) to 793 mg/kg (2008), which is somewhat lower than the mean calculated from combined Canadian and international data. The Rasmussen *et al.* (2001; 2008) studies revealed that, in Ottawa, ON, household dust had a different multi-element signature than outdoor soil and dusts, and may be found at higher concentrations compared to their concentrations in residential garden soil.

Rasmussen *et al.* (2001; 2008) reported that house dust can contribute significantly to exposure to metals in residential urban environments. Rasmussen *et al.* (2008) compared the median bioaccessible and median total zinc concentrations (in particles of settled dust, sieved to less than 150 µm) in both house dust and residential garden soils and found median total zinc concentrations of 626 µg/g in house dust and 84 µg/g in garden soil, and median bioaccessible zinc concentrations of 410 µg/g in house dust and 18 µg/g in garden soil. When the fraction of bioaccessible zinc was compared to total zinc, 63 per cent of total zinc was found to be bioaccessible in house dust compared to 25 per cent in soil. Bioaccessibility was found to be fairly constant across size fractions (i.e. 60–68 per cent) and organic carbon appeared to explain approximately 58 per cent of the overall variability of zinc bioaccessibility (Rasmussen *et al.* 2008). Rasmussen *et al.* (2008) also noted that the elevated indoor/outdoor (I/O) ratios in this study were comparable to reported I/O ratios in other urban residential settings in the United Kingdom and New Zealand (Culbard *et al.* 1988; Kim and Fergusson 1993). No statistical correlations were observed between indoor dust concentrations and outdoor soil.

Based on 12 studies (see Appendix III), a mean indoor air concentration of 1,278 ± 6,649 mg/kg (mean ± SD) was estimated as representative of indoor settled dust concentrations in Canada (HC 2011a) for the purposes of developing the inhalation EDI for zinc, used in the human health SQG calculation. The EDI methodology is further dicussed in Section 7.2 of this document.

2.5.4 Soil

Zinc concentrations measured in Canadian soil are summarized in Appendix I.

Zinc in soil can be attributed to the weathering of rock and soil parent materials, atmospheric deposition (e.g., erosion, volcanic eruptions, burning of coal and other fossil fuels, mining, smelting of non-ferrous metals), agricultural use of sewage sludge, and the use of agrochemical fertilizers and pesticides.

Total zinc content in soil is largely dependent on the composition of the parent rock material (Kiekens 1990). An average earth zinc crustal abundance of 70 mg/kg has been reported (Kabata-Pendias 2011; Committee on Medical and Biological Effects of Environmental Pollutants [CMBEEP] 1979), and worldwide average soil concentrations generally range from approximately 10 to 300 mg/kg (He *et al.* 2005). In soils, the average is 90 ppm with a range of 1-900 ppm (Adriano 2001).

Deposition of atmospheric zinc on soil can occur as wet or dry deposits. Higher levels of deposition occur in areas with an elevated atmospheric source of zinc pollution, and these processes are dependent on particle size, as paritcles of smaller aerodynamic diameter can travel long distances (WHO 2001).

The largest anthropogenic emitter of zinc, by far, is the mining and smelting industry. Other important sources are fertilizers and atmospheric fallout followed by lesser yet still significant contributions from wastes of various origins (agricultural, sewage sludge, fly ash, etc.) (Nriagu and Pacyna 1988 in WHO 2001). Proximity to roadways is also associated with higher concentrations of zinc in soil originating from tire debris (WHO 2001; ATSDR 2005).

It is acknowledged that zinc concentrations in soil throughout Canada will vary based on geology and anthropogenic inputs; however, a single background soil concentration is required to develop soil quality guidelines for use throughout Canada. The Canadian background concentration of zinc in soil was developed from GSC data obtained from till samples (<63µm) that were analyzed by AAS/ICP-ES following aqua regia digestion (partial digestion by HCl and HNO_3) (Grunsky 2010a; Rencz *et al.* 2006). These samples were obtained from the following regions: Newfoundland/Labrador, New Brunswick, Québec, Nunavut, Northwest Territories, Manitoba, Saskatchewan, Alberta and British Columbia. A background soil concentration of 48.1 (±48.4) mg/kg was calculated from 7,398 samples of glacial till from across Canada. This data was also used to determine the estimated daily intake for Canadians (HC 2011a).

Soil data from surface layers are subject to greater influence from anthropogenic activities and may better represent those to which ecological and human receptors are exposed. A study in the St. Lawrence Valley, QC found that the rates of atmospheric deposition to soil and sediment were highest in the spring and fall, and lowest in the winter and summer, with deposition rates generally decreasing in recent years due to lower emissions to the atmosphere brought about by regulations (Gelinas *et al.* 2000). The following provides more information on regional and geological variations in zinc concentrations.

Friske *et al.* (2014) present data from the 2008 and 2009 North American Soil Geochemical Landscape Project Field Surveys. The data includes zinc concentrations in the public health layer (0–5cm) in addition to the 0–30cm layer, A, B and C horizons. Data are available for different analytical techniques: near total digestion (4-acid digestion), ICP-MS/AES; partial/aqua regia digestion, ICP-MS/AES; Instrumental Neutron Activation Analysis (INAA) and water leach, ICP-MS/AES. Data are also available for different fractions: <2mm fraction milled and unmilled and the <63µm unmilled. Bioaccessibility data were not included in this study. It should be noted that data are not available for each depth by all listed analysis types and fraction sizes (Table 2 of Friske *et al.* 2014 outlines the data available). While zinc found in the surface soil layers is more relevant to humans and many other organisms, the sample size for this data is much smaller, therefore it was not chosen at this time for the derivation of the SQG. The zinc concentrations (median values) for the <2mm fraction (not milled), analyzed by ICP-MS/AES following

partial/aqua regia digetion are as follows: public health layer, 60 mg/kg (n=363); 0–30cm, 48 mg/kg (n=157); A horizon, 60 mg/kg (n=329); B horizon, 53 mg/kg (n=336); C horizon, 49 mg/kg (n=355) (Friske *et al.* 2014). For the B and C horizons, the <63µm unmilled fraction was analyzed by ICP-MS/AES following partial/aqua regia digestion. Zinc concentrations were found to be 66 mg/kg (B horizon, n=335) and 60 mg/kg (C horizon, n=356) (Friske *et al.* 2014).

The North American Soil Geochemical Landscape Project (NASGLP) Maritime soil survey data are available for the public health layer as well as A, B and C horizons (<2mm milled fraction) for New Brunswick, Nova Scotia and Prince Edward Island. From these data, a mean concentration of 51.5 mg/kg (SD=37.5, n=184) was calculated for the public health layer. The mean concentration calculated was 56.3 mg/kg (SD=39.1, n=156) for the A horizon, 64.5 mg/kg (SD=33.6, n=183) for the B horizon and 67.6 mg/kg (SD=34.5, n=176) for the C horizon (Grunsky 2010b).

The clay fraction contains the majority of zinc followed by the silt and sand fractions. Zinc content in Ontario soils was related to soil texture, with sandy soils (40 mg/kg) lower than loam (64 mg/kg), clayey (62 mg/kg) and organic soils (66 mg/kg) (Adriano 2001).

In 1993, the Ontario Ministry of the Environment and Climate Change (formerly the Ontario Ministry of the Environment and Energy) analyzed surface soil samples (0–5 cm) from old urban and rural parklands throughout Ontario—that were not affected by local point sources of pollution—for a wide variety of chemicals to determine average background concentrations known as "Ontario Typical Range" (OTR_{98}=97.5 percentile of the distribution). A 2011 OTR update (Ontario Ministry of the Environment [OMOE] 2011a) includes additional soil data gathered since the 1993 document was produced. The OTR_{98} concentrations in rural parkland (n=101 sites in 1993; n=110 in 2011) and old urban parkland soils (n=60 sites in 1993; n=97 in 2011) were 120 and 140 mg/kg, respectively, in 1993 and 160 and 180 mg/kg, respectively, in 2011. However, this Ontario dataset contains samples from the Sudbury area, where metal mining and smelting are common, that increase the upper percentiles and the mean (OMEE 1993; OMOE 2011a). Soil from the Sudbury and Rouyn-Noranda regions may have average zinc concentrations as high as 360 mg/kg in contaminated areas (Feisthauer *et al.* 2006) while organic soil from an uncontaminated site 60 km away from metal smelters in Québec showed a concentration of 100 mg/kg (Johnson and Hale 2004).

Agricultural soils from southwestern Ontario measured by Whitby *et al.* (1978) had average zinc concentrations of 88, 87 and 71 mg/kg in the A, B and C horizons, respectively. Frank *et al.* (1976) sampled soils collected from all agricultural areas of Ontario and reported an average zinc content in the plough layer soil of 56.7 mg/kg. Organic soils contained the highest average concentration (66.3 mg/kg), while sandy soils contained the lowest average concentration (39.9 mg/kg).

Zinc in the surface horizons of northwest Alberta agricultural soils was reported to be 55 mg/kg by Soon (1994) and 94 mg/kg by Soon and Abboud (1990), with a subsurface soil (20–35 cm) concentration of 81 mg/kg. In southeast and central Alberta background soils ranged in concentration from 29 to 235 mg/kg (Dudas and Pawluk 1980). All A horizons were enriched with zinc in comparison to the levels determined in respective C horizons (Dudas and Pawluk 1980), with the clay fraction containing the majority of zinc followed by the silt and the sand fractions.

Samples collected from control sites in an agricultural area in Saskatchewan in 2003 had mean concentrations ranging from 74 to 107 mg/kg (Lipoth and Schoenau 2007).

Background soil concentrations in Mt. Robson Provincial Park, BC were found to be 15.5–137 mg/kg (Arocena *et al.* 2006). A regional zinc concentration of 94.1 mg/kg was measured in a non-impacted area of Trail, BC (locations chosen to avoid influence from smelters in the vicinity) (Sanei *et al.* 2007).

2.5.5 Surface Water

Zinc concentrations in Canadian water are summarized in Appendix I.

The main contributors of anthropogenic zinc to water are mining and manufacturing processes (production of zinc, iron, chemicals, pulp and paper and petroleum products), followed by domestic and urban waste water and sewage sludge in roughly equal amounts (Nriagu and Pacyna 1988 in WHO 2001; ATSDR 2005) and atmospheric fallout (ATSDR 2005). With a frequency of detection of 95 per cent, the concentration of zinc from urban runoff ranges from 10-2,400 µg/L across 19 American cities (Cole *et al.* 1984). Deposition from air accounts for a significant amount of zinc inputs to the surface water of the Great Lakes (Superior: 60–80 per cent; Erie and Ontario: 20–70 per cent) (Eisenreich and Strachan 1992; Nriagu 1986; Strachan and Eisenreich 1988), amounting to 21.80-56.50 µg/m^2 per year in 1989–90 (Flegal *et al.* 1989 in Nriagu *et al.* 1996).

Areas close to pollution sources can contain high levels of zinc in surface water, for example, Mann *et al.* (1989 in WHO 2001) found an average zinc concentration of 900 µg/L in acidic mine tailing ponds in Canada. However, ambient aquatic levels of dissolved zinc in Canada are usually lower than 40 µg/L (Eisler 1993), with fresh water generally containing higher concentrations than marine waters. The typical global seawater concentration range is 0.0005–0.1 µg/L (WHO 2001), with lower concentrations at the surface than at depth and decreasing with distance from shore. However, lower concentrations of zinc (and some other metals such as lead and cadmium) can also be found in some freshwater systems (Nriagu *et al.* 1996). Similar levels have been measured in the United States (ATSDR 2005), where the majority of concentrations are below 50 µg/L (range: 20–50 µg/L). Surface waters from non-contaminated areas exhibited concentrations that ranged from 1.6-4.4 µg/L from Ontario lakes (Shuhaimi-Othman *et al.* 2006) and below 1 µg/L (0.09-0.3 µg/L) in Lakes Superior, Erie and Ontario (Nriagu *et al.* 1996). Doyle *et al.* (2003) report a Canadian background zinc level for surface waters of 12 µg/L from the 95th percentile; however, the methodology is not provided. The surface layer (epilimnion) of offshore Great Lakes waters becomes zinc-depleted in the summer months (Nriagu *et al.* 1996).

Lakes near anthropogenic zinc inputs in the Sudbury, ON region (near nickel smelters) were found to have average zinc concentrations similar to other background levels (2.2–18 µg/L) (Shuhaimi-Othman *et al.* 2006; Pyle *et al.* 2005).

The levels of zinc found in surface water are not included in the determination of the EDI.

2.5.6 Groundwater

Zinc concentrations in Canadian groundwater are summarized in Appendix I.

The Ontario Ministry of the Environment (OMOE 2011a) provides background groundwater values from wells not affected by known contamination through two data sources: the Provincial Groundwater Monitoring Network (PGMN) (data for 2002–2007) and the Drinking Water Surveillance Program (DWSP) (data for 1997, 1999–2002). The PGMN 97.5 percentile background for zinc is 159 µg/L (n=419) and the DWSP 97.5 percentile background value is 42.06 µg/L (n=747).

2.5.7 Drinking Water

The Canadian aesthetic drinking water objective for zinc is ≤5.0 mg/L (15,000 µg/L). At higher concentrations, water tends to be opalescent, develops a greasy film when boiled and tends to have an astringent taste (HC 1987, reaffirmed 2005 [HC 2014)]).

There is considerable variation in reported concentrations of zinc in water, particularly between treated and untreated water. Tap water may contain higher levels of zinc compared to the supply source, due to zinc leaching from galvanized or copper alloy distribution pipes or plumbing components. After one overnight stand in the systems, levels up to 100 times greater than that of treated water were found (HC 1987). Thus, in Gananoque, ON the concentration of zinc after treatment was less than 0.01 mg/L in 1986, whereas the mean concentration at seven stations after an overnight stand was 0.309 mg/L (range: 0.03–1.17 mg/L) (HC 1987). Sharrett *et al.* (1982) reported zinc levels in tap water from Seattle, Washington homes with galvanized pipe to be about 10 times higher than from homes in the same location with copper pipe. The water type (soft or hard) can also influence the concentration of metals; however, zinc was found to be evenly distributed over all ranges of hardness in a Canadian tap water study (Neri *et al.* 1975).

The data that were used to develop the EDIs for the Canadian population for drinking water were those provided by the DWSP (data for 1998–2008) (OMOE 2010), Saskatchewan Ministry of Environment (2009) (data for 2000–2009) and the Water Resources Management Division of the Government of Newfoundland and Labrador (2010) (data for 2000–2009). The calculated arithmetic mean zinc concentration in drinking water using data from the three provinces is 0.0111 ± 0.0491 mg/L with a range of 0.0001 mg/L to 2.861 mg/L, (n=14,714) (HC 2011a). The EDI values are used in the human health SQG calculation. The EDI methodology is further discussed in Section 7.2 of this document.

Although zinc concentrations in bottled water were not considered in the calculation of the EDI, Health Canada's Chemical Health Hazard Assessment Division, Food Directorate has analyzed various samples to determine zinc levels in several bottled waters sold in Canada. In a Canada-wide survey of various drinking water sources (Dabeka *et al.* 2002), the mean concentration of zinc in tap water was found to be 0.0022 mg/L. Mean concentrations measured in bottled water were 0.0081 mg/L for mineral water, 0.012 mg/L for spring water, 0.00349 mg/L for distilled water and 0.0016 mg/L for soda water.

2.5.8 Sediment

Zinc concentrations in Canadian sediments are summarized in Appendix I.

The sources of zinc in sediments are numerous, as bed sediment is formed from settled suspended sediment originating from erosion, runoff, atmospheric deposition, liquid effluent and transport from all atmospheric, terrestrial and aquatic sources. There is also some indication that sediments can release zinc to surface waters following spring overturn of the water column (Nriagu *et al.* 1996).

In Canada, baseline sediment concentrations are approximately 90 mg/kg, with higher concentrations associated with sediments close to point sources and occasionally in natural deposition zones ([National Research Council] NRC 1979). Data from the National Geochemical Reconnaissance program, started by the GSC in 1973, reveals that mean zinc concentrations in Canadian freshwater lake sediments range from 83.1 mg/kg in Newfoundland to 120 mg/kg in the Northwest Territories, and mean zinc concentrations in freshwater stream sediments range from 60.8 mg/kg in Ontario to 397 mg/kg in the Northwest Territories (EC 1999b). Background zinc concentrations measured in sediment cores from the Fraser River Basin ranged from 86.6 to 181.9 mg/kg (Gallagher *et al.* 2004). Stream zinc sediment concentrations were measured at 140 mg/kg in the Yukon (Gamberg *et al.* 2005).

Documented sediment concentrations greater than 300 mg/kg (dry weight) have been detected in fluvial lakes of the St. Lawrence River (Desrosiers *et al.* 2008), the Great Lakes (Marvin *et al.* 2007) and the Sudbury region in Ontario (Shuhaimi-Othman *et al.* 2006) due to the discharge of sewage and/or industrial effluents. In other studies, concentrations from the Great Lakes were found to range from 80.7 to 114 mg/kg (Gewurtz *et al.* 2008) and an average of 66 mg/kg in the Bay of Fundy (Hung and Chmura 2007).

Contamination from mining, smelting and other industries can result in elevated lake and river sediment zinc concentrations. Mean concentrations of zinc contaminated sediments ranged from 313 mg/kg to 3,100 mg/kg, and a maximum concentration of 26,250 mg/kg, with notably affected areas including Lake Ontario, Flin Flon and Rouyn-Noranda lakes (EC 1999b).

For more background and contaminated site zinc sediment data for freshwater and marine environments, see the scientific supporting document entitled *Canadian Sediment Quality Guidelines for Zinc* (EC 1999b).

2.5.9 Biota Used as Human Food

Zinc is an essential element, and the dietary sources of zinc vary widely. Zinc is abundant in red meats, certain seafood and whole grains. Zinc levels tend to be higher in animals than in plant products and are particularly high in some shellfish, especially oysters (National Institute of Health 2011). Shellfish samples from Canada (1991–2001) had a range of 28–129 mg/kg dry weight (Allen-Gil *et al.* 1997; Chou *et al.* 2003). Freshwater fish sampled between 1991 and 2003 from Canadian lakes showed a much wider range of values—3.1–1,441 mg/kg dry weight—however the maximum value was found in perch liver caught in a lake in the Sudbury region and the vast majority of measurements were well below 100 mg/kg for all other lakes (Allen-Gil *et al.* 1997; Evans *et al.* 2005; Morrisey *et al.* 2005; Muir *et al.* 2005; Pyle *et al.*

2005). Wild game caught in Canada between 1995 and 2003 had a range of values similar to shellfish: 34.9–192.7 mg/kg dry weight (Gamberg *et al.* 2005; Parker and Hamr 2001; Parker 2004; Pedersen and Lierhagen 2006). The year-to-year range for mean zinc values in Canadian western red spring wheat was from 32.7 to 34.9 mg/kg dry weight between 1996 and 1998 (Gawalko *et al.* 2002). The Canadian total diet study conducted in Vancouver in 2007 measured zinc levels (wet weight) of 0.0054 mg/kg in marine fish, 0.0049 mg/kg in freshwater fish and 0.011 mg/kg in shellfish (Dabeka 2007). Foods consumed in a traditional arctic diet in Canada, involving 102 species of animals and plants, were found to contain zinc concentrations ranging from 0.01 to 1.5 mg/kg (Kuhnlein *et al.* 2002).

2.5.10 Commercial Food

HC's Food Directorate provided estimated daily intakes of zinc that were derived from the HC Total Diet Study (TDS) (Dabeka *et al.* 2010). For the most recent TDSs, food samples were analyzed using ICP-MS and concentrations were reported in ng/g (fresh weight). The detection limits of the analyses varied depending on the type of food and the reagent blanks. Separate intake rates in µg per day were provided for various age groups and for each year from 2000 to 2007, inclusive. Data from the 2000–2007 Canadian TDSs were used in the determination of the EDI of zinc from food sources. The data were provided directly by HC's Food Directorate and then normalized to correspond to the age groups used in this document. Zinc concentrations measured in food in Canada are summarized in Appendices I and II. The EDI methodology is further discussed in Section 7.2 of this document.

Worldwide concentrations of zinc in common foods were determined in a recent review (Scherz and Kirchoff 2005). The concentrations ranged from 3.1 to 4.45 µg/g in milk, 0.01 to 0.049 µg/g in meat, 0.0031 to 0.026 µg/g in fish, 0.0234 to 0.0447 µg/g in cereal, 0.0015 to 1.3 µg/g in vegetables, trace to 0.64 µg/g in fruit and 1.21 to 3.66 µg/g in nuts. Similar zinc concentrations have been measured in Canadian total diet studies from 2003 to 2007 (HC 2011b); the highest concentrations measured were in seeds (48.27–54.69 µg/g); rice and bran cereal (26.80–49.89 µg/g); meat, poultry and eggs (12.08–82.11 µg/g); herbs and spices (10.77–19.77 µg/g); shellfish (9.60–11.47 µg/g); and milk-based formula (6.46–10.88 µg/g).

Galvanized cooking and storage vessels are a potential source of zinc in food. Acidic foods can dissolve zinc metal that may subsequently be converted to zinc salts and ingested. However, the refining and preparation of foods can also result in a decrease in zinc content (HC 1987). Because zinc is mainly located in the germ and bran portions of grains, as much as 80 per cent of the total zinc is lost during milling. Many breakfast cereals are fortified with zinc (Institute of Medicine [IOM] 2001).

2.5.11 Breast Milk

In humans, zinc concentrations are higher in the colostrum than in mature milk and decline with the length of lactation (Wasowicz *et al.* 2001). The IOM (2001) reported breast milk concentrations varying between 4.7 mg/L at 7 days post-birth and 0.8 mg/L at 9 months.

Zinc concentrations in breast milk from lactating mothers in Newfoundland had mean concentrations that decreased from 4.58 mg/L at 1 week to 1.14 mg/L at 12 weeks (n=19 mothers with full-term infants), and from 5.31 mg/L at 1 week to 1.17 at 12 weeks (n=24 mothers with pre-term infants) (Friel *et al.* 1999).

Thirteen studies from different countries were identified to develop the EDI for zinc in human breast milk (Abdulrazzaq *et al.* 2008; Almeida *et al.* 2008; Casey *et al.* 1985; Coni *et al.* 1990; 2000; daCosta *et al.* 2002; Honda *et al.* 2003; Kosanovic *et al.* 2008; Krachler *et al.* 1998, 1999; Li *et al.* 1989, 1990; Matos *et al.* 2009). Based on these studies, the mean concentration in human breast milk was determined to be 3.58 ± 3.33 mg/L (n=1199) (HC 2011a). These data were used to develop the zinc EDI values for exposure to human breast milk (infants). The EDI methodology is further dicussed in Section 7.2 of this document.

2.5.12 Consumer Products

Numerous medical, cosmetic and dietary supplement products contain zinc. Dietary supplements of marine origin obtained in 2004 to 2005 were found to have zinc concentrations ranging from 5.1 to 50 mg/kg (Leblond *et al.* 2008). In Canada, zinc dosage in supplements is regulated at <50 mg/day for adults and associated with associated copper requirements to allow for a 25:1 Zn:Cu ratio to prevent potential copper deficiency in the presence of zinc (HC 2007). Zinc is also found in tobacco products (Jenkins 1986). High zinc levels in children have been associated with chewing metal toys made with zinc alloys (Mitchell 1995). Individuals that smoke or use zinc supplementation will have a greater exposure to zinc than the general population (ATSDR 2005). High levels of zinc in the range of 53 to 440 mg/kg have been found in coal (Finkelman 1999) and in paint (31.10 mg/kg) (Mielke *et al.* 2001). Zinc compounds can also be found in products such as sunscreens, deodorants and anti-dandruff shampoos (HC 2010a).

Consumer products were not included in the EDI calculation.

2.6 Existing Criteria and Guidelines

Existing guidelines, criteria, or standards for zinc in soil from provincial, national and international agencies are summarized below in Table 2.

Table 2: Existing Soil, Groundwater, and Surface Water Quality Criteria, Guidelines and Standards for Zinc from Various Jurisdictions

Jurisdiction	Category	Criterion/Guideline	Reference
Ontario	Surface soil and groundwater standards in a potable groundwater situation		OMOE 2011b
	Agricultural	340 µg/g	
	Residential/parkland	340 µg/g	
	Industrial/commercial	340 µg/g	
	Groundwater (all land uses)	1,100 µg/L	
	Surface soil and groundwater standards in a nonpotable groundwater situation		

	Agricultural	340 µg/g	
	Residential/parkland	340 µg/g	
	Industrial/commercial	340 µg/g	
	Groundwater (all land uses)	1,100 µg/L	
	Subsurface soil standards in a potable groundwater situation		
	Residential/parkland	(24,000)[i] 15,000 µg/g	
	Commercial/industrial	(24,000) 15,000 µg/g	
	Subsurface soil standards in a nonpotable groundwater situation		
	Residential/parkland	(24,000) 15,000 µg/g	
	Commercial/industrial	(24,000) 15,000 µg/g	
Québec	Generic criteria for soils and groundwater		Ministère de l'environnement du Québec 1998
	Soil A (background concentrations)	110 mg/kg (St-Lawrence lowlands)	
	B (moderate soil contamination requiring additional study)	500 mg/kg	
	C (threshold value requiring immediate cleanup)	1,500 mg/kg	
	Groundwater Drinking water	5,000 µg/L	
	Seepage into surface water	67 µg/L	
Yukon	Numerical Soil Standards (Human Health Protection)		Yukon Government, Department of Environment 2002
	Agricultural	10,000 µg/g	
	Residential/urban park	10,000 µg/g	
	Commercial	30,000 µg/g	
	Groundwater (potable)	150 to 15,000 µg/L (soil pH and land use dependent)	
Alberta	Alberta Tier 1 Soil Remediation Guidelines (fine and coarse soils)		Alberta Environment and Parks 2016
	Natural Area	200 mg/kg	
	Agricultural	200 mg/kg	
	Residential/Parkland	200 mg/kg	
	Commercial	360 mg/kg	
	Industrial	360 mg/kg	
British Columbia	Matrix Numerical Soil Standards (Human Health Protection)		BC Ministry of Environment 2009
	Agricultural	10,000 µg/g	
	Residential/urban park	10,000 µg/g	
	Commercial	30,000 µg/g	
	Groundwater	150 to 15,000 µg/L (soil pH and land use dependent)	
Canada	Canadian Water Quality Guidelines (Aquatic Life)	30 µg/L	Canadian Council of Resource and Environment Ministers 1987
	Canadian Water Quality Guidelines (Irrigation)	1,000-5,000 µg/L	
	Canadian Water Quality Guidelines (Livestock)	50,000 µg/L	
Canada	Canadian Soil Quality Guidelines		CCME 1999
	Agricultural	200 mg/kg	

	Residential/parkland	200 mg/kg	
	Commercial	360 mg/kg	
	Industrial	360 mg/kg	
European Commission	Predicted no effect concentration for the terrestrial compartment ($PNEC_{add, terrestrial}$) based on microbe-medaited processes	27 mg/kg	European Commission 2008
	Predicted no effect concentration for the terrestrial compartment ($PNEC_{add, terrestrial}$) based on invertebrates and plants	26 mg/kg	
Netherlands	Groundwater target values and soil and grandwater intervention values		Ministry of Infrastructure and Environment 2009
	Target value shallow groundwater (<10m)	65 µg/L	
	National background concentration for groundwater (>10m)	24 µg/L	
	Target value groundwater deep (>10m)	24 µg/L	
	Intervention value soil	720 mg/kg	
	Intervention value groundwater	800 µg/L	
USA	Ecological Soil Screening Levels		US EPA 2007a
	Plants	160 mg/kg	
	Soil invertebrates	120 mg/kg	
	Avian wildlife	46 mg/kg	
	Mammalian wildlife	79 mg/kg	

[l] Ontario Standards bracketed number applies to medium- and fine-textured soil; non-bracketed number applies to coarse-textured soil.

3.0 ENVIRONMENTAL FATE AND BEHAVIOUR

3.1 Air

Zinc primarily enters the atmosphere bound to particulate matter in an oxidized state via several natural and anthropogenic processes including: wind erosion of soils and industrial materials, burning of coal, oil, or sewage sludge and the refining of zinc and other metals. Due to its boiling point of 907°C (Table 1), zinc is unlikely to volatilize in the environment, except under extreme conditions such as during volcanic activity and forest fires (Richardson *et al.* 2001). Global natural emissions (45,000 tonnes) to air are mainly due to windborne soil particles (19,000 tonnes), volcanoes (9,600 tonnes) and forest fires (7,600 tonnes) (Nriagu 1989). Another natural source is sea salt spray (600 tonnes) (Adriano 2001; Nriagu 1989).

Transport and deposition of atmospheric zinc vary according to the size of particles (which depends on the emission source), the properties of the zinc compounds concerned and meteorological conditions. Zinc is removed from the atmosphere by dry and wet deposition (wet dominating with 60–90 per cent of the deposition), and it can be transported over long distances from both natural and anthropogenic sources (Steinnes and Friedland 2006); however, studies conducted in the vicinity of smelters have documented that the deposition of atmospheric zinc generally occurs within 25 kilometres of smelters (Hopkin 1986; Ma *et al.* 1983; Storm *et al.*

1994). Anthropogenic inputs via atmospheric loading of zinc to soils can readily occur, even in remote areas, resulting in soil concentrations above natural background (Allen-Gil *et al.* 1997).

Much of the zinc released from industrial processes is adsorbed to particles in the respirable range of aerodynamic diameters. Zinc in urban and industrial areas was present on particles with diameters of up to 5 µm. One study reported that 73 per cent of zinc was in particles smaller than 4.7 µm in samples taken near smelters (WHO 2001).

There is no estimate for the atmospheric half-life of zinc, but the fact that zinc is transported over long distances indicates that its lifetime in air is at least a few days (ATSDR 2005). The rates of atmospheric deposition are highest in the spring and fall and lowest in the winter and summer, with rates generally decreasing in recent years due to lower emissions to the atmosphere (Gelinas *et al.* 2000).

3.2 Water and Sediment

The largest natural source of zinc to surface water can be attributed to soil erosion. As a result of soil weathering, soluble zinc compounds are formed and released to water (WHO 2001). Zinc concentrations in fresh water depend significantly on local geological influences and anthropogenic inputs (WHO 2001).

In ocean water, the vast majority of dissolved zinc is complexed to organic ligands (Bruland *et al.* 1991 in Nriagu *et al.* 1996). In freshwater systems, zinc is present in an oxidation state of +2 and can be found in several chemical forms including hydrated ions, dissolved chemical species and inorganic and organic complexes (ATSDR 2005). Zn^{2+} predominates in water, where organic matter content determines which chemical form of zinc is present (WHO 2001). Relative mobility of zinc in water (and soil) is mainly determined by the solubility of the compound, salinity and pH (ATSDR 2005).

Phosphates and iron hydroxides play an important role in the transfer of heavy metals from river water to sediments (Houba *et al.* 1983 in WHO 2001). Soluble inorganic zinc compounds may be present as a small fraction of total zinc. Soluble inorganic zinc (e.g., zinc choride, zinc sulphate) hydrolyzes to form zinc hydroxide, which precipitates readily (Patterson *et al.* 1977). Zinc may also co-precipitate with hydrous oxides of iron or manganese (US EPA 1979 in ATSDR 2005). The natural buffering capacity of water will generally prevent these reactions from causing a significant decrease in pH (WHO 2001).

Zinc and other heavy metals are highly partitioned to suspended and bed sediments, and most of the zinc introduced into aquatic environments is eventually deposited in the sediments (WHO 2001). Zinc will precipitate in polluted waters with high zinc concentrations, particularly at pH >8 (US EPA 1979 in ATSDR 2005). The pH levels influence zinc's complex stability. Dissolution of zinc increases at low pH, low hardness and high temperature. Desorption of zinc increases with salinity (Helz *et al.* 1975 in ATSDR 2005), probably due to higher concentrations of alkali and alkaline cations. Partitioning coefficients for zinc in suspended solids in water (log Kp) have been estimated to range from 3.43 to 4.48 for North American rivers (Bockting 1992 in European Union 2008), and from 3.5 to 6.9 from the literature (n=47) (Allison and Allison 2005).

Acidic waters (pH of 3.1 and 5.1) were found to have lower concentrations of sorbed zinc than pH neutral waters (Gunderson and Steinnes 2003) as non-colloidal inorganic compounds such as zinc carbonate, zinc hydroxyl carbonate or zinc silicate (WHO 2001). Alkaline fresh waters contained mostely bound zinc associated with organic and inorganic particles (WHO 2001). Computer simulations undertaken relative to an industrial channel (Lachine canal) with elevated metal concentrations in Montréal predicted that zinc would be present mainly as $Zn(OH)_2$, Zn^{2+} and $ZnOH^+$ in the water column. The relative abundance of these species was influenced by the presence or absence of solid phases. When solid phases were allowed to precipitate (at high pH values), Zn^{2+} was the predominant species and $Zn(OH)_2$ concentrations were reduced by 29 per cent. Although Zn^{2+} remained the major species in waters, it was also predicted that, after disturbance from dredging in assigned theoretical conditions, $ZnSO_4$ and $ZnCl^+$ could be major species in the water column (Degtiareva and Elektorowica 2001). Reimer and Duthie (1993) found a negative correlation between sulphate and zinc levels in the sediments of water bodies in the Sudbury and Muskoka, ON regions. They noted that in these areas of high sulphur input, unbuffered lakes become more acidic, increasing the solubility of zinc in the aquatic environment.

The predominant species in anoxic sediments is zinc sulphide (Casas and Crecelius 1994 in WHO 2001), whereas zinc will generally be associcated with hydrous iron and manganese oxides in the upper oxic layer of sediment (US NAS 1977 in WHO 2001).

In seawater, zinc chemistry is dependent on the concentrations of dissolved organic complexing agents. Increases in dissolved and suspended zinc were noted in estuarine zones where a five-fold increase in leachable zinc was noted in sediments compared to freshwater zones (Grieve and Fletcher 1977 in WHO 2001).

3.3 Soil

Zinc in soil is distributed between five fractions: dissolved in pore water, exchangeably bound to soil particles or organic ligands, occluded in secondary clay minerals and insoluble metal oxides/hydroxides, and adsorbed or complexed in primary minerals (EU 2008). The majority of zinc in soil will be in the adsorbed portion rather than in soluble forms in pore water or soil solution (WHO 2001). Atmospheric inputs of zinc to soil almost always exceed output due to leaching and biomass production (Kabata-Pendias 2011). The mobility of zinc in soil is dependent on the specific compound and is affected by factors such as soil pH, clay mineralogy, soil organic matter content, rainfall and infiltration, and soil drainage, with pH being the main factor (Hesterberg 1998; Impellitteri *et al.* 2003; WHO 2001). Soil texture is closely related to zinc concentrations, with the lowest values associated with sandy soils (Kabata-Pendias 2011). Zinc may precipitate as a hydroxide at pH values greater than 8.0. It may also form stable organic complexes that can affect the mobility and/or solubility of zinc, as well as form immobile precipitates such as zinc sulphide under anaerobic conditions. Zinc becomes more soluble as pH decreases; therefore, zinc is more mobile at low pH (<5) (Duquette and Hendershot 1990). As a result, leaching of zinc occurs more readily from sandy and acidic soils.

Zinc is highly reactive in soils, so that, in addition to inorganic Zn^{2+}, zinc is present as part of both soluble and insoluble organic compounds. In a study of 66 contaminated soils from North America and Europe, 60–80 per cent of zinc in the soil samples was found to be bound to

organic matter (Stephan *et al.* 2008). Within a given soil, equilibrium exists between the different forms of zinc (adsorbed, exchangeable, secondary minerals, insoluble complexes) in the liquid and solid soil phases. Plant uptake, losses by leaching, input of zinc in various forms, changes in the moisture content of the soil, pH changes, mineralization of organic matter and changing redox potential of the soil will influence the equilibrium. Due to the complexity of zinc interactions in soil, zinc transport behaviour in soil cannot be predicted accurately and soil adsorption effects cannot be separated from solution effects such as precipitation (WHO 2001). The variability of zinc adsorption to soil was demonstrated by Stephan *et al.* (2008), who found that soil pore water partitioning coefficients range from 17 to 13,100 L/kg. A summary by Allison and Allison (2005) of soil-water partition coefficients from the literature (n=21) shows a much larger range: log K_d of -1.0 to 5, with a median value of 3.1. Another study of Dutch soils found soil pore water partitioning coefficients to vary from 6 to 6,762 L/kg, with some correlation to pH (Janssen *et al.* 1997a). Several researchers have presented estimates of zinc in soil solution relative to total zinc concentration in soil. In a review on zinc behaviour, Lindsay (1972) estimated that 2–10 per cent of total soil zinc is present in the soil solution. In a review by Kiekens (1990), soil solution zinc concentrations were estimated to range from 3×10^{-8} to 3×10^{-6} M.

The actual speciation of zinc is difficult to predict (Meers *et al.*, 2006). Under typical conditions, most zinc is adsorbed onto soils. However, at some contaminated sites, zinc may be less strongly adsorbed (i.e., more readily extracted) than naturally occurring zinc, such as seen in old wood impregnation sites in Finland (Schultz *et al.* 2004). Rodriguez *et al.* (2008) found zinc in mine tailings samples to be mainly in an acid-extractable form, while background zinc in soils from nearby areas was mainly in the residual form (non-silicate bound metal). Diesing *et al.* (2008) investigated the exchangeability and speciation of zinc in six soils polluted with metals from various sources. They found that available zinc was present in smelter-impacted soils primarily as weakly bound octahedral zinc, but that the exchangeability of weakly bound octahedral zinc to Zn^{2+} in solution decreased with increased soil pH and organic matter.

EDTA-extractable zinc was reported to decrease with depth in the profile (Lindsay 1972). Zinc is easily adsorbed by mineral and organic components, thus its accumulation in the surface horizon was also observed (Kabata-Pendias 2000). Zinc associated with smaller soil particles can be more available (mobile) than zinc associated with larger soil particles (Preciado and Li 2006).

Bioavailability of zinc compounds is variable and dependent on the physical-chemical properties of the zinc species and the surrounding environment. As mentioned above, soil texture, pH, organic matter content and various other factors can influence the bioavailability of zinc (Amaral *et al.* 2006). The bioavailability of zinc in contaminated urban soils was found to be lower without industrial sludge amendments used for agriculture than in sludge-amended soils (Bose and Bhattacharyya 2008). That study found that zinc bioavailability from sludge-amended soils was 9.9 to 21 per cent. The bioavailability of zinc in soil may also decrease with time as zinc gets adsorbed to soil and forms complexes with the surrounding material.

3.4 Indoor Settled Dust

Rasmussen *et al.* (2008) reported that indoor dust and soil are geochemically distinct. Indoor dust has approximately five times the organic matter as soil samples (Rasmussen *et al.* 2008). Organic

carbon is a key factor controlling metal partitioning and bioavailability. The higher metals concentration in indoor dust compared to soils may be explained by the affinity some metals have for organic matter, in addition to the smaller particle size of dust (Rasmussen *et al.* 2008). Consequently, the use of outdoor soil metals data to predict indoor dust concentrations may result in the underestimation of indoor dust exposures (Rasmussen 2004). In areas where spatial correlations do exist between indoor dust and exterior soil concentrations, the correlations suggest an elevated external source (i.e., mining or other sources of industrial contamination) (Rasmussen *et al.* 2004).

4.0 BEHAVIOUR AND EFFECTS IN BIOTA

The available information on the toxicological effects of zinc on soil microbial processes, terrestrial plants and invertebrates, mammals and birds has been reviewed and summarized in this chapter in support of the derivation of environmental soil quality guidelines. This information has been tabulated in Appendices VIII to XV.

The lowest observed effects concentration (LOEC) endpoints reported in the toxicity tables (Appendices VIII to XV) represent the LOEC at which there was a statistically and biologically significant difference from the controls, as reported by the author(s). If no such statistical tests were reported by the author(s), the percentage of adverse effect, as compared to the controls, from zinc concentrations within the soil will be calculated from the data presented by the author(s). This percentage of adverse effect is represented by an effective concentration (EC_x) endpoint within the toxicity tables. Actual EC_x endpoints reported by the author(s), such as EC_{25} or EC_{50}, will be presented as such without any calculation of a percentage of adverse effect. Measured concentrations and metal extraction methods are reported in the toxicity tables only if they involve digestion of soil with a strong acid, such as HCl or HNO_3. Otherwise, the nominal concentrations are reported.

4.1 Soil Microbial Processes

Toxicity studies consulted for soil microbial processes are presented in Appendix VIII, while studies selected for use in soil quality guidelines derivation are listed in Appendix XIII. Soil enzyme activities reported here were not included in the selected data since they may not represent measured effects of chemicals on soil microbial populations. This is due to the fact that many enzymes produced by plants and microbes can exist and function extracellularly in soil for varying periods of time, depending on soil micro-environmental factors (Tabatabai 1982).

The toxicity of zinc to microbial processes may be affected by characteristics of the soil. In particular, the background concentration of zinc in the soil has been shown to have significant effects, with higher background concentrations resulting in more tolerance of the microbial processes to zinc contamination, likely due to the microbial communities being adapted to higher zinc concentrations (Broos *et al.* 2007; Smolders *et al.* 2004). Some studies have shown high pH to decrease the toxicity of zinc to nitrification processes, while others have not found a relationship between pH and toxicity to microbial processes. Soil cation exchange capacity has also been shown to influence the effects of zinc on soil respiration.

Carbon mineralization decreased by 21 per cent after 8 weeks of treatments at a concentration of 10 mg Zn/kg in a sandy soil of pH 4.9 (Cornfield 1977). This author also reported that a concentration of 100 mg Zn/kg reduced the amount of CO_2 release by 45 per cent relative to the controls. In a similar soil texture but at a higher pH of 6.0, Bhuiya and Cornfield (1972) documented a 16 per cent inhibition of carbon mineralization at 1,074 mg Zn/kg.

Bhuiya and Cornfield (1974) studied the effect of a single concentration of zinc, added as zinc oxide, on nitrogen mineralization at various pH values in a sandy soil. At a concentration of 1,074 mg Zn/kg in soil, no effect on nitrogen mineralization was observed at pH 6.0 while 8 per cent and 32 per cent reductions in nitrogen mineralization were recorded at pH 7.0 and 7.7, respectively.

Doelman and Haanstra (1984) measured the effects of relatively high zinc concentrations upon soil respiration, in soils of varying texture and pH. In a sandy soil with a pH 7.0, a 44 per cent inhibition of respiration was observed at 1,000 mg Zn/kg. In a silt loam soil of pH 7.7 and higher clay (19 versus 2 per cent) content, soil respiration was inhibited by 38 per cent at a concentration of 8,000 mg Zn/kg. Lower inhibition rates of soil respiration (26 per cent) were also documented by these authors in clay (pH 7.5) and sandy peat at 3,000 mg Zn/kg.

Many studies on nitrification monitored an inhibition over time in various soil types with added zinc. In a single concentration study, Bhuiya and Cornfield (1974) measured a 13 per cent inhibition of nitrification in sand at a concentration of 1,074 mg Zn/kg at pH 7.0, a 33 per cent decrease at pH 7.7 and no effect at pH 6.0. The level of nitrification decreased with time, suggesting an adaptation of the microbial population to zinc concentrations in soil. Liang and Tabatabai (1977) monitored zinc effects on nitrification in soils ranging in texture from loam to silty clay and in pH from 5.8 to 7.8, and reported similar inhibition levels (12–15 per cent) after 20 days at 327 mg Zn/kg. The inhibitive effects of zinc on nitrification after 10 days in various soils ranged from 39 to 72 per cent at 327 mg Zn/kg (Liang and Tabatabai 1978). Wilson (1977) obtained a greater inhibitive response on the nitrification process with various soil types. A 70 per cent inhibition occurred in a sandy loam soil, pH 6.2, after 3 weeks of treatment with 100 mg Zn/kg. The same zinc concentration in a loamy sand of pH 7.4 resulted in 27 per cent inhibition of nitrification. Wilson (1977) also observed complete inhibition of nitrification in sandy loam, loamy sand and clay loam soils after 7 weeks of treatment with 1,000 mg Zn/kg soil.

Bollag and Barabasz (1979) studied the effects of various zinc nitrate concentrations on the process of denitrification. In a 21-day exposure period, a 40 per cent reduction in denitrification was observed at 250 mg Zn/kg soil in a silt loam soil of pH 6.75. Under similar test conditions, a 65 per cent reduction in denitrification was documented at 500 mg Zn/kg soil.

Chaudri *et al.* (1992) monitored the long-term effects of zinc on nitrogen fixation by *Rhizobium leguminosurum* over time. In a sandy loam soil, pH 6.5, nitrogen fixation was not affected after two months at 455 mg Zn/kg soil. However, 18 months of treatment at 385 mg Zn/kg soil resulted in complete inhibition of nitrogen fixation.

Elevated zinc concentrations in soil were also found to inhibit enzyme activity. Doelman and Haanstra (1986) monitored urease activity in soils at varying zinc levels over 6-week and 18-month periods. For the sand, sandy loam and clay soils tested, the EC_{50} decreased with time, ranging from 420 to 1,780 mg Zn/kg after 6 weeks and ranging from 90 to 290 mg Zn/kg after 18 months. These authors also determined LOEC values for urease activity (10 per cent reduction)

ranging from 30 to 460 mg Zn/kg at 6 weeks and from 1 to 160 mg Zn/kg at 18 months. In another study, phosphatase activity was inhibited by 28 to 59 per cent in loam to clay loam soils treated with 1,643 mg Zn/kg (Juma and Tabatabai 1977). Ohya *et al.* (1985) investigated glucose mineralization in a sandy clay loam soil at 1,000 mg Zn/kg and reported a 44 per cent inhibition of activity after 24 hours and an 11 per cent decrease after 96 hours. These authors also observed an increase in bacterial population in the zinc-amended soil after 48 hours and suggest that the population had increased by selection for zinc tolerance.

4.2 Terrestrial Plants

4.2.1 Metabolic Fate and Behaviour

Zinc availability to terrestrial plants is a function of soil physico-chemical properties and plant biological characteristics (OMEE 1993; Tyler *et al.* 1989). The uptake rate of zinc by plants generally increases with increasing zinc concentration in soil (Chang *et al.* 1983; Nwankwo and Elinder 1979; Petruzelli *et al.* 1989; Schuhmacher *et al.* 1993; Smith 1994). Uptake and distribution of zinc in higher plants is influenced by the form of zinc (Davis-Carter and Schuman 1993; Mortvedt and Giordano 1975; Speaker 1991; Wallace 1963); while zinc is primarily available to plants as Zn^{2+}, organically complexed zinc may also be taken up by the roots (Broadley *et al.* 2007). Other factors affecting uptake include other metal ions present in the system (Fontes and Cox 1993; Sarkunan *et al.* 1989; Wallace 1989; Wallace and Berry 1989), soil phosphorus levels (Grant and Bailey 1989; Hamilton *et al.* 1993; Singh 1992; Smilde *et al.* 1974), cation exchange capacity, soil texture (Chang *et al.* 1983; Singh 1992), soil properties such as pH (Davies 1992; Schuhmacher *et al.* 1994; Smith 1994; van der Watt *et al.* 1994; Xian and Skohohifard 1989) and organic matter content (Hamilton *et al.* 1993; Pierzynski and Schwab 1993; Singh 1992). Plant species (Bernhard *et al.* 2005; Chino and Chino 1991; Chukwuma 1993; Sieghardt 1990; Soon 1994; Tyler *et al.* 1989; Vedagiri and Ehrenfeld 1991; Viets *et al.* 1954), intraspecies variations (Nriagu 1980; Yang 1994), the developmental stage of the plant (McKenna *et al.* 1993; Sanka and Dolezal 1992), presence of mycorrhizae (Faber *et al.* 1990) and growth conditions (Markert and Weckert 1989) such as temperature, light and nutrient availability are all contributing factors to the interaction between zinc and plants. One study conducted on soils contaminated with filter dust from a brass foundry found that ageing did not appear to affect plant uptake of zinc (Hilber *et al.* 2007). It has been suggested that the zinc concentration in the soil solution may be indicative of the zinc available for uptake by plants (Meers *et al.* 2006).

Zinc may accumulate significantly in fine roots and other rhizosphere components due to more acidic conditions and the increased ability to form organic complexes (Belling Abler 2004; Courchesne *et al.* 2006); humic acid has been shown to increase zinc adsorption to the plant root surface and uptake by plants (Kalis 2006). Zinc in plants may also, to some extent, be redistributed to the soil (Page *et al.* 2006). Zinc is transported to the shoot along with oxygen atoms, either as a metal-organic acid complex or as a hydrated cation, and accumulates in cell vacuoles (Salt *et al.* 2002).

Data on the accumulation of zinc in terrestrial plants are summarized in Appendix XVI. Zinc concentrations in plants have been shown to vary seasonally in some studies, with the highest

concentrations measured during the winter and the lowest during the spring, likely due to dilution by plant growth (Deram *et al.* 2006). Due to the ability of plants to regulate zinc, uptake relative to the soil concentration may be reduced at high soil concentrations (Almas *et al.* 2006); studies of soil contaminated by smelter flue dust found that concentrations in plants versus soils followed the plateau (saturation) model for describing relationship between Zn plant content (y) and soil Zn content (x), $y = ax^b$, with 'b' being less than 1 (Dudka *et al.* 1996). A review of several bioaccumulation studies (Efroymson 2004) found a statistically significant relationship between the natural logarithm of the zinc concentration in soil and the natural logarithm of the concentration in plant foliage. Notten *et al.* (2005), however, found zinc concentrations in plant leaves to be poorly correlated with concentrations in soil.

Some plants, referred to as zinc hyperaccumulators, can tolerate and accumulate large amounts of zinc. Zinc hyperaccumulators may not regulate zinc uptake, however, and zinc concentrations as high as 40,000 mg/kg have been measured in shoot tissue from hyperaccumulators, compared to typical concentrations in the range of 20 to 100 mg/kg in most plants (Guerinot and Eide 1999). Zinc concentrations in these plants are often higher in plant shoots than in the roots (Knight *et al.* 1997).

4.2.2 Toxicity

A summary of consulted zinc toxicity studies for plants is presented in Appendix IX. Appendix XIV summarizes the selected toxicity data used on plants and invertebrates for the derivation of the soil quality guideline.

The toxicity of zinc to plants is likely related to the uptake of zinc, as well as plant tolerance. Other factors may also influence zinc toxicity; for example, one study found that increasing the supply of calcium and magnesium reduced the toxicity of zinc to sugar beet seedlings (Saleh *et al.* 1999).

Data for the acute toxicity (exposure period less than 14 days) of zinc to terrestrial plants are available for the effect on seedling emergence of lettuce (*Lactuca sativa*) and radish (*Raphanus sativus*) (EC 1995). For radish planted in an artificial soil, ranging in pH from 4.0 to 4.2 and in organic matter content from 4.7 to 6.3 per cent, a 50 per cent reduction in seedling emergence was observed at concentrations ranging from 280 to 670 mg Zn/kg soil. The no observed effect concentration (NOEC) ranged from 100 to 230 mg Zn/kg soil under similar test conditions. A 50 per cent reduction in seedling emergence of lettuce was documented at concentrations ranging from 400 to 720 mg Zn/kg soil while the NOEC ranged from 200 to 250 mg Zn/kg soil in artificial soils of pH 4.0 to 4.2 and organic matter contents of 4.7 to 10.4 per cent.

Chronic toxicity data (exposure period greater than 14 days) of zinc effects are available; an 18 per cent yield reduction, measured as total dry matter weight, in onion grown for eight weeks occurred at 400 mg Zn/kg in a clay loam soil, pH 8.3 (Dang *et al.* 1990). Smilde *et al.* (1992) measured a 53 per cent reduction in the yield of endive grown to maturity in a sandy soil (pH 4.2) at 60 mg Zn/kg soil and a 91 per cent yield reduction at 80 mg Zn/kg. In the same study spinach exhibited lower sensitivity to zinc than endive with a 27 per cent yield reduction at 80 mg Zn/kg. No effect on spinach yield was observed on spinach grown to maturity in a loam soil (pH 7.2) at 160 mg Zn/kg.

Sheppard *et al.* (1993) measured various responses of lettuce (*Lactuca sativa*) and turnip (*Brassica rapa*) in several soil types with differing zinc concentrations. In a sandy soil with pH 6.3, 50 per cent reductions in first bloom and seed yield were observed for turnip at 25 mg Zn/kg and a 50 per cent reduction in seedling emergence occurred at 65 mg Zn/kg. Lettuce grown in an identical sandy soil was less sensitive to zinc, with a 50 per cent reduction in seedling emergence at 207 mg Zn/kg. When grown in a clay garden soil of pH 7.3, no effect on seedling emergence of lettuce or turnip was observed at 1,000 mg Zn/kg, the highest concentration used, while 50 per cent reductions in first bloom and seed yield were noted in turnip at 600 and 715 mg Zn/kg soil, respectively. In a silty clay soil (pH 7.9), no response was observed on seedling emergence of lettuce at 1,000 mg Zn/kg, the maximum applied concentration, while turnip exhibited 50 per cent reductions in emergence, first bloom and seed yield at 600 mg Zn/kg.

MacLean (1974) studied the effects of zinc sulphate on plant yield in sandy soils. Corn (*Zea mays*) grown over six weeks in a fine sandy loam (pH 4.9) demonstrated a 13 per cent yield reduction at 303 mg Zn/kg soil, while no effect on yield was reported for sandy loam soils, pH 7.2 to 7.5, with 329 mg Zn/kg soil. MacLean (1974) documented 100 per cent mortality of lettuce (*Lactuca sativa*) tested at 303 mg Zn/kg soil over 5 weeks in a fine sandy loam soil (pH 4.9). Alfalfa grown in this soil over 16 weeks exhibited a 71 per cent reduction in yield at 303 mg Zn/kg soil. As with corn, no effect on the dry matter yield of lettuce or alfalfa was observed at 329 mg Zn/kg soil for the sandy loam soils, pH 7.2 and 7.5, respectively.

Jones (1983) and Jones *et al.* (1988) studied the yields of agricultural crops grown in well drained drumlin soils of pH 7.1, sampled within one metre of hydroelectrical transmission towers in Ontario. Levels up to 1,425 mg Zn/kg soil were measured originating from the corrosion of the galvanized towers. However, no effects were noted on the yields of lettuce or radish grown in this soil for 45 days (Jones 1983) or on corn yield grown to maturity (Jones *et al.* 1988). Mortvedt and Giordano (1975) also documented the effect of zinc sulphate on corn yield. In a sandy loam soil of pH 5.5, a 50 per cent reduction in corn (*Zea mays*) yield was observed at 240 mg Zn/kg soil. This study also reported 100 per cent mortality at a concentration of 1,400 mg Zn/kg soil.

Blackgram (*Vigna mungo*) grown for 65 days in soils of pH 6.2 exhibited a 22 per cent yield reduction at 200 mg Zn/kg soil and a 45 per cent yield reduction at 250 mg Zn/kg (Kalyanaraman and Sivagurunathan 1994). Another study documented yield reductions of wheat and rice occurring at much higher zinc concentrations (Muramoto *et al.* 1990). Wheat grown for 23 weeks in an alluvial soil exhibited a 64 per cent yield reduction at 1,000 mg Zn/kg, an 82 per cent yield reduction at 10,000 mg Zn/kg and no grain yield at 30,000 mg Zn/kg. For rice grown in this alluvial soil, a 25 per cent yield reduction occurred at 50,000 mg Zn/kg (Muramoto *et al.* 1990).

The effects of zinc on trees grown in sandy soils were documented in several studies. Jack pine (*Pinus banksiana*) grown in a sandy loam soil of pH 6.0 demonstrated 25 per cent reduced root yields at 25 mg Zn/kg and 6 per cent decreased shoot yields at 50 mg Zn/kg over a 12 week treatment (Dixon and Buschena 1988). These authors reported white spruce root and shoot yield decreases of 13 per cent and 28 per cent, respectively, at 50 mg Zn/kg soil. Hagemeyer *et al.* (1993) grew beech (*Fagus grandifolia*) saplings for two years in a soil mixture of sand, peat and forest soil of pH 4.8 with various zinc levels. At 65 mg Zn/kg soil, the thickness of tree growth rings demonstrated a 50 per cent growth reduction, and shoot growth was reduced by 39 per cent.

Mortality of all beech trees occurred at 490 mg Zn/kg soil after the first year. Hogan and Wotton (1984) grew black spruce (*Picea mariana*) and jack pine in sandy loam to loamy sand soils of pH 4.9. No effects on the concentration of other foliar nutrients were noted at 1,200 mg Zn/kg.

4.3 Terrestrial Invertebrates

4.3.1 Metabolic Fate and Behaviour

Earthworms are important organisms in the soil macrofauna since their activities mix the soil, improving aeration, water permeability and mineral turnover in the soil. Earthworms may be an important component of terrestrial food chains by providing a food source for many small mammals and birds (Honda *et al.* 1984). Earthworms accumulate zinc and are therefore useful bioindicators of soil zinc contamination (Ma 1982; Ma *et al.* 1983). There is some evidence in the literature that earthworms are able to physiologically regulate the concentration of zinc in their tissues, resulting in lower bioconcentration factors at higher soil concentrations (Ireland 1979; Morgan and Morgan 1988; Panda *et al.* 1999).

Soil characteristics play a significant role in the uptake of zinc by worms (Boyd and Williams 2003). Earthworms access mainly exchangeable zinc, such as that associated with soil pore water, rather than residual zinc associated with soil particles or soil organic matter (Scott-Fordsmand *et al.* 2004). Ma (1982) found that the level of zinc in the earthworm *Lumbricus rubellus* was generally related to zinc concentration in the soil and highly correlated with zinc concentrations in low pH soils. At lower pH, the soil adsorbs less zinc, therefore increased concentrations in the soil solution occur, rendering zinc more bioavailable to earthworms. Ma *et al.* (1983) reported a negative correlation between cation exchange capacity (CEC) and zinc concentration in earthworms: the concentration in the worms increased as CEC decreased. This effect was also attributed to an increase in bioavailable zinc as CEC decreased. Organic matter content in the soil did not affect zinc uptake by earthworms (Ma 1982). Aluminum oxyhydroxides and soil clay content may also have some effect on bioaccumulation of zinc by earthworms (Janssen *et al.* 1997b). One study found that the amount of zinc bioavailable to earthworms could be approximated by the $Ca(NO_3)_2$-extractable zinc concentration (Conder and Lanno 2000).

In a study on woodlice (*Porcellio scaber*), zinc was found to accumulate within the body tissues without any positive correlation with zinc levels in the leaf litter and soil (Hopkin 1986). Mortality of woodlice occurred at a zinc concentration of 1,430 mg Zn/kg in leaf litter (Hopkin 1986). A long-term study conducted by Hopkin and Hames (1994) over 360 days for woodlice found mortality of all individuals at a concentration of 1,090 mg Zn/kg maple leaf litter. A predator of woodlice, the spider *Dysdera crocata* was found by Hopkin and Martin (1985) to accumulate large amounts of zinc in its body with no ill effects.

A study conducted using ground beetles that were fed prey with high zinc concentrations found that the zinc concentration in the beetles remained relatively constant during and after exposure, indicating that zinc was regulated (Kramarz 1999a). Conversely, the zinc concentration in centipedes was found to increase over time during exposure and decrease when exposure ended (Kramarz 1999b).

In heavily contaminated areas, the ingestion of plant material may be the most significant source of zinc exposure for some species; one study measured zinc concentrations in soil, plant leaves and snails, and found the zinc concentration in snails to be correlated mainly with the concentration in plant leaves, with a less significant relationship to soil concentrations (Notten *et al.* 2005).

4.3.2 Toxicity

The consulted data for the effects of zinc on invertebrates are summarized in Appendix X. Appendix XIV summarizes selected toxicity data on plants and invertebrates used for the derivation of the soil quality guideline.

The toxicity of zinc to soil invertebrates may be affected by various factors. In particular, the bioavailability of zinc is known to affect its toxicity. Ageing can decrease the bioavailability of zinc, particularly at high pH levels, resulting in lower toxicity in aged soils compared to freshly spiked soils (Lock and Janssen 2003a). One study found that the toxicity of zinc to earthworms increased at higher temperatures (Khan *et al.* 2007), although another study found no clear effect of temperature on zinc toxicity to collembola reproduction (Sandifer and Hopkin 1997).

Malecki *et al.* (1982) looked at the toxic effects of different chemical forms of zinc (acetate, carbonate, chloride, nitrate, oxide and sulphate) on the growth and reproduction of young earthworms (*Eisenia foetida*) over eight weeks. The metals were mixed with a known quantity of horse manure, which was placed on top of screened soil. LOECs for cocoon production and body weight ranged from 500 to 4,000, while for body weight, the LOEC values ranged from 2,000 to over 40,000. Generally, reproduction was a more sensitive endpoint for various zinc compounds than growth.

Several studies documented zinc mortality to earthworms on various soil types. Sheppard *et al.* (1993) determined an LC_{50} of 80 mg Zn/kg for *Eisenia foetida* in clay soil, pH 7.3. However, in sandy soil of pH 6.3, the LC_{50} was determined at 460 mg Zn/kg and in silty clay (pH 7.9), the LC_{50} was 600 mg Zn/kg. Environment Canada (1995) reported slightly higher soil zinc concentrations, which resulted in 50 per cent mortality for earthworms. The LC_{50} ranged in concentration from 700 to 800 mg Zn/kg soil when *E. foetida* were exposed over 14 days to zinc chloride in artificial soil of pH 4.0 to 4.2. Under similar test conditions, this study documented 25 per cent earthworm mortality for soil concentrations of 500 to 700 mg Zn/kg and no effect on mortality for concentrations ranging from 300 to 500 mg Zn/kg.

Neuhauser *et al.* (1985) documented an LC_{50} of 662 mg Zn/kg for earthworms exposed for 14 days to zinc nitrate in an artificial sandy loam soil (pH 6.0). Spurgeon *et al.* (1994) also conducted 14-day LC_{50} tests for earthworms on an artificial sandy loam soil with zinc nitrate and reported an LC_{50} of 1,010 mg Zn/kg soil. Under the same experimental conditions, an exposure period of 56 days resulted in an LC_{50} of 745 mg Zn/kg, and a NOEC for mortality of 289 mg Zn/kg. A 50 per cent reduction in cocoon production occurred after 56 days at 276 mg Zn/kg, while the estimated NOEC for cocoon production was 199 mg Zn/kg soil.

Van Gestel *et al.* (1993) studied the effects of various concentrations of zinc on growth and reproduction of *Eisenia andrei* in an artificial soil. Significant effects included reduced reproduction (31 per cent) and increased production of malformed cocoons (89 per cent) at 560

and 1000 mg Zn/kg soil, respectively. These authors also found that earthworms had some ability to regulate their body content of zinc. However, concentrations of soil zinc exceeding 1,000 mg Zn/kg soil did cause an increase of zinc body content in earthworms. Hartenstein *et al.* (1981) also reported the effects of various soil zinc concentrations on the growth of earthworms. This study reported LOECs ranging from 1,300 to 13,000 mg Zn/kg soil for earthworms (*Eisenia foetida*) in silt loam soils of pH 6.5 to 7.0 when exposed for eight weeks to soil supplemented with zinc sulphate.

Juvenile garden snails (*Helix aspersa*) fed food spiked with zinc experienced significant adverse effects to growth (4-week EC_{50} = 5,800 µg Zn/g food) (Gomot-De Vaufleury 2000).

Some studies have suggested that zinc may have additive toxicity with other trace elements, including cadmium, copper and lead (Posthuma *et al.* 1997).

4.4 Mammals and Birds

4.4.1 Metabolic Fate and Behaviour

Zinc is absorbed from the intestine according to the needs of the animal and is primarily excreted in the feces. Absorption is dependent on the animal species and is influenced by factors such as age, dose and length of exposure (Davies *et al.* 1977; Eisler 1993; Ott *et al.* 1966). Many studies document the accumulation of absorbed zinc in the liver and kidneys of sheep, cattle, poultry and rats (Dewar *et al.* 1983; Llobet *et al.* 1988; Ott *et al.* 1966). In a study of zinc-amended diets, Llobet *et al.* (1988) also found significant increases of zinc concentrations in the heart, bone and blood tissues of rats.

Mammals and birds obtain zinc primarily from dietary sources. In a review by NRC (2005), the following values for zinc content in various animal feeds were reported: pasture, 17-60 ppm; cereal grains, 20-30 ppm; soybean meal, 50-70 ppm dry weight. For humans, 77 per cent of zinc intake is obtained from dairy products, meat, fish, poultry, grains and cereals (ATSDR 2005). Livestock may ingest elevated levels of zinc by licking galvanized or painted surfaces or by ingestion of contaminated soil, vegetation or water. Birds may ingest elevated zinc by ingestion of zinc shot or by ingestion of contaminated food sources such as vegetation, insects or other prey.

Inorganic salts of zinc, including zinc oxide, carbonate, acetate, chloride and metallic zinc are readily available sources for mammals. Those salts that are insoluble are solubilized by gastric juice. Contamination of food, water and soil with large amounts of zinc can occur from storage in galvanized containers, deposition of zinc from mining activities and by corrosion of galvanized structures such as electrical transmission towers (Jones 1983; NRC 2005; Nriagu 1980).

Grazing sheep tested on Zn-contaminated lands near smelters were found to have much higher zinc concentrations in their liver and kidneys than control sheep grazing on uncontaminated lands (Reif *et al.* 1989). Strong correlations occurred between soil zinc concentrations and the level of zinc accumulated in organs. Trowbridge's shrews (*Sorex trowbridgii*), deer mice (*Peromyscus maniculatus*) and shrew-moles (*Neurotrichus gibbsii*) collected in a sludge-treated

forest in Washington State, USA, accumulated zinc in their kidneys and liver with no other observed effects (Hegstrom and West 1989). Some birds have the ability to eliminate zinc when returned to a normal-level diet after an extended dosage of high zinc. Dewar *et al.* (1983) found that zinc concentrations in the liver of laying hens fed a normal diet after exposure to a short-term high zinc diet returned to almost normal levels after six weeks.

Zinc is not normally believed to biomagnify in terrestrial food chains, although some evidence of increasing zinc concentrations with higher trophic levels has been observed in arctic seabirds (Borga *et al.* 2006).

Puls (1988) found that there is a strong relationship between zinc and calcium (Ca) in the dietary requirements of cattle. The recommended daily requirement for cattle is 45 mg Zn/kg dry matter intake with 0.3 per cent Ca. For each additional 0.1 per cent Ca, 16 mg Zn/kg should be added to the diet.

4.4.2 Toxicity

Appendices XI, XII and XV summarize available toxicological data of the effects of zinc on mammals and birds.

Zinc toxicity has been reported in livestock with the common exposure routes being galvanized feed troughs, galvanized wire, feed supplements with high zinc contents, heavy use of zinc-containing fertilizers and fungicides, and the direct ingestion of zinc contaminated soils and forage. Zinc toxicity has been observed in many animals, but its effects are so diverse that it is difficult to identify any single mechanism as being responsible for death (Campbell and Mills 1979; Ott *et al.* 1966). Clinical signs of zinc toxicity include loss of appetite, decreased water consumption and dehydration, increased mineral consumption, loss of condition (decrease in weight gain or loss of weight), weakness, jaundice, diarrhea and paralysis of the legs in birds (Allen *et al.* 1983; Dean *et al.* 1991; Gasaway and Buss 1972; Ott *et al.* 1966). Morphological changes noted as a result of zinc ingestion included: anemia; lesions in the kidney, gizzard and pancreas; reduction in gonad growth of young ducks; enlarged and pale kidneys; significant increase of zinc in liver, kidneys, heart, bone and blood tissues; decreased copper and increased iron concentrations in the liver; effect on kidney function; and pathological changes in the pancreas, kidney, liver, rumen, abomasum, small intestine and adrenal gland.

At zinc doses of 33.6 mg Zn/kg body weight (BW)/day for four weeks in milk-substitute diets, reduced rates of weight gain were observed in lambs (Davies *et al.* 1977). Mallard ducks fed single zinc shots in gelating caplets showed reduced weight gain at a calculated dose of 17.9 mg Zn/kg BW (French *et al.* 1987). Young mallard ducks exhibited an average 19 per cent weight loss at 109 mg Zn/kg BW and moderate to severe weight loss (23–45 per cent) at 158 mg Zn/kg BW when given in their feed over 60 days (Gasaway and Buss 1972). Food consumption and the rate of weight gain both decreased with increasing zinc dosage (Dewar *et al.* 1983; Gasaway and Buss 1972; Ott *et al.* 1966). At 178 mg Zn/kg BW, Ott *et al.* (1966) observed that lambs stopped feeding completely and reduced their water intake by 75 per cent compared to controls. Weight loss and reduced food intake were also observed in a 28-day study of 1-day-old chicks at a dosage of 1,074 mg Zn/kg BW (Dean *et al.* 1991). No effects on weight gain or food intake were observed in rats up to a dosage of 640 mg Zn/kg BW (Llobet *et al.* 1988).

The effects of zinc exposure on kidneys, liver and pancreas were documented for mammals and birds in several studies. Mallard ducks developed liver and kidney lesions at 17.9 mg Zn/kg BW (French *et al.* 1987). In young poultry, gizzard and pancreatic lesions were observed at dosages greater than or equal to 65.7 mg Zn/kg BW (Wight *et al.* 1986). Rats developed kidney lesions and exhibited renal dysfunction at 320 mg Zn/kg BW when exposed for three months (Llobet *et al.* 1988). Decreased copper content in the livers of sheep were noted at 33.6 mg Zn/kg BW (Davies *et al.* 1977).

Zinc fed to 7-week-old mallard ducks at 109 mg Zn/kg BW for a period up to 60 days resulted in leg paralysis, yellowish to reddish-yellow kidneys, moderate to high reduction in gonadal growth and mortality (Gasaway and Buss 1972). In the same study, a dosage of 158 mg Zn/kg BW resulted in leg paralysis along with reduction in gonadal growth and mortality of all ducks within 40 days (Gasaway and Buss 1972).

Reproductive effects of zinc on pregnant sheep were reported by Campbell and Mills (1979). When pregnant sheep were fed 20 mg Zn/kg BW during the first 10 days of the gestation period and 10 mg Zn/kg BW during the final 10 weeks, 64 per cent of the offspring was non-viable. These authors also observed reduced rates of weight gain and feed consumption by the sheep and lower offspring weights when pregnant sheep were fed zinc. Dewar *et al.* (1983) studied the effects of high zinc diets on laying hens and reported that a diet of 25,000 mg Zn/kg prevented hens from laying eggs. Zinc is often used in commercial egg production to control and improve egg laying (Dewar *et al.* 1983; Eisler 1993; Wight *et al.* 1986).

4.5 Essentiality

4.5.1 Terrestrial Plants

Zinc is an essential element for normal plant growth, forming an essential component of many enzymes (Brennan 1992; Giordano and Mortvedt 1980; Guerinot and Eide 1999; Nable and Webb 1993; Salt *et al.* 2002; Soper *et al.* 1989; Wallace and Berry 1989). Zinc is commonly deficient during the growth of agricultural crops, especially in high pH soils (Broadley *et al.* 2007; Elinder 1986; Guerinot and Eide 1999). Terrestrial plants predominantly absorb zinc as Zn^{2+} from the soil solution, but hydrated zinc and several other complexes and organic chelates may also be absorbed (Kiekens 1990). Most soils contain sufficient total zinc levels for plant growth but plant uptake is dependent upon the availability, solubility and movement of zinc to plant roots (Eisler 1993; Giordano and Mortvedt 1980; Soon and Abboud 1990). The amount of zinc in soil must satisfy plant growth requirements while not exceeding concentrations that cause phytotoxicity to plants or that have the potential to contaminate other organisms along the food chain. Zinc deficiency in plants can result in stunted growth, interveinal chlorosis and leaf symptomatologies such as small leaves, malformations and dieback (US EPA 2007a).

4.5.2 Terrestrial Invertebrates

Zinc is an essential element for invertebrates, and internal invertebrate concentrations are generally regulated within a narrow range (Lock *et al.* 2000). Currently, there is inadequate

published data to determine the minimum levels required to prevent zinc deficiency in invertebrate species.

4.5.3 Mammals and Birds

Zinc is present in all tissues and is an essential trace element for proper growth, development and function in mammals and birds (NRC 2005). It also has an important regulator role in the immune system, which has demonstrated atrophy in species that were deficient (Donmez *et al*. 2003). It has been reported that more than 300 metalloenzymes require zinc in which the metal is located at the active site of the enzyme and is involved in their catalytic activity (IOM 2001). Zinc assures stability of biological molecules such as DNA and RNA and of biological structures such as membranes and ribosomes (Underwood 1971). Zinc is an inducer of metallothioneins, proteins that temporarily store zinc and aid in counteracting zinc toxicity (NRC 2005). Zinc is also involved in carbohydrate and protein metabolism, as well as synthesis of indoleacetic acid (US EPA 2007a). Zinc requirements for young domestic animals and fowl range from about 40 to 100 ppm in the diet. A review of zinc dietary requirements by the US EPA (2003) included the following species: cats (15 mg/kg diet), cattle (9–14 mg/kg diet), chickens (35–40 mg/kg diet), ducks (60 mg/kg diet), mink (59–66 mg/kg diet), mice (10 mg/kg diet), pheasant (60 mg/kg diet), rats (12–25 mg/kg diet), sheep (17–33 mg/kg diet) and turkeys (40–70 mg/kg diet).

Zinc is considered an essential trace element in humans by nutritionists (IOM 2001). Insufficient levels of zinc in the diet can produce reversible functional or structural abnormalities associated with biochemical changes. However, excessive levels of zinc may have adverse effects. For essential trace elements, there is a safe range of intakes between deficiency and toxicity that is represented by a U-shaped dose-response curve (HC 2009) (Sections 5.1 and 5.8).

5.0 BEHAVIOUR AND EFFECTS IN HUMAN AND MAMMALIAN SPECIES

5.1 Overview

Zinc is an essential element and is important in human nutrition as it plays a role in the function of more than 300 enzymes (IOM 2001; US EPA 2005). Zinc-containing proteins and enzymes are involved in many aspects of the chemical processing of genetic material, including both replication and translation. The primary biological functions of zinc are catalytic, structural and regulatory (IOM 2001). Zinc deficiency in humans can result in dermatitis, anorexia, growth retardation, poor healing, hypogonadism and impaired immune function (ATSDR 2005). At high doses, zinc can interfere with the status of other essential elements such as iron and copper and cholesterol levels. Very high doses have been associated with serious neurological disease. Decreased copper status has been determined to be the critical effect in humans for the determination of the Upper Limit UL by IOM (2001). A discussion of zinc deficiency is not provided, as it is not relevant for the purpose of deriving a soil quality guideline for zinc.

The mammalian toxicology of zinc has been recently reviewed by various health agencies that include European Food Safety Authority (EFSA 2014), ATSDR (2005), US EPA (2005), IOM

(2001) and WHO (2001). It is not the role or the intention of this document to critically re-evaluate the mammalian toxicology of zinc. This has already been done by agencies responsible for protecting human health in Canada and other jurisdictions. Because of the extensive data available on zinc toxicity, summaries of human studies will be emphasized in the following sections. However, due to observational limitations and study performance limitations, some critical health tissue- or organ-level effects from animal studies may not be captured in human studies. Accordingly, the sections below represent a summary of the key studies that have been used for the development of human health for zinc, as relevant to this document.

As was discussed earlier, zinc may be found in a variety of chemical forms at contaminated sites including: zinc chloride; zinc oxide (zincite), zinc sulphate and zinc sulphide (sphalerite and wurtzite). However, as mentioned in Chapter 3, the dominant chemical species for zinc in water and in soil is Zn^{2+}. Therefore, the toxicological review presented here is thus related to zinc in its elemental form (Zn^{2+}).

5.2 Classification

HC (2009), ATSDR (2005) and the Netherlands' National Institute of Public Health and the Environment (RIVM, 2001) do not classify zinc as carcinogenic. US EPA's Integrated Risk Information System (2005) states that *"there is inadequate information to assess carcinogenic potential of zinc, because studies of humans occupationally-exposed to zinc are inadequate or inconclusive, adequate animal bioassays of the possible carcinogenicity of zinc are not available, and results of genotoxic tests of zinc have been equivocal."* The International Agency for Research on Cancer (IARC) has not classified zinc for carcinogenicity. In 2008 the European Union risk assessment report on zinc chloride reported that zinc supplementation or deficiency could influence carcinogenisis through inhibition or promotion actions, but no clear evidence is available (European Union 2008). At the current time, no major health agency considers zinc to have carcinogenic potential.

Considering the relevant environmental exposure pathways, the relative toxicities of different salts and the available toxicological data, it was determined that the toxicological reference values for zinc should be based on oral exposure to zinc in its elemental form. The most toxic species of zinc is Zn^{2+}, due to this species being more bioavailable than other species. Neither the US EPA nor ATSDR have developed inhlation reference values for zinc (ATSDR 2005; US EPA 2005). Inhalation has only been shown to cause route-specific detrimental health effects after exposure to high levels of zinc vapour from welding, mainly in occupational settings (US EPA 2005). Although this SQG has been developed based on an oral toxicity reference value (TRV), inhalation and dermal exposures are also included as they contribute to the overall zinc body burden.

5.3 Toxicokinetics

5.3.1 Absorption

5.3.1.1 Oral Exposure

In humans, the bioavailability of zinc and related compounds is complex and is influenced by factors such as the physical and chemical speciation, route of exposure, absorption and retention, interactions with other elements and diet. Zinc is not generally present in foods in free ion form. Zinc is primarily absorbed by the small intestine through both passive diffusion and a saturable carrier-mediated process (IOM 2001; US EPA 2005). The fraction absorbed is difficult to determine as zinc is also secreted into the gut and re-absorbed, and absorption is dependent on the zinc status of the individual (HC 1987). Based on the results of seven studies, ATSDR (2005) indicated that absorption in humans ranged from 8 to 81 per cent following short-term exposure to zinc supplements and that differences were likely due to the amount of zinc ingested and the amount and type of food consumed. ATSDR (2005) noted that people with adequate nutritional zinc levels would typically absorb 20–30 per cent of ingested zinc but that can be higher when people have fasted (i.e., 68–81 per cent). Under conditions of deficiency, uptake may increase to approximately 90 per cent (Hunt *et al.* 2008),

Many regulatory and dietary factors have been reported to influence zinc absorption; of these, one of the most important is phytate (myoinositol hexaphosphate). Phytic acid, particularly when ingested with calcium, reduces zinc absorption by forming insoluble precipitates. Therefore, the high phytate content of foods such as cereals and legumes reduces the bioavailability of zinc.

The amount and type of protein in the diet impacts the efficiency of zinc absorption. At near neutral pH, zinc binds strongly to proteins and less digestible proteins (such as proteins in cow's milk or plants) also lead to lower absorption (IOM 2001).

Zinc and iron are known to interact. The type of interaction depends on the levels of each metal and how they are administered. Decreased zinc absorption was observed when iron was administered in water with an iron:zinc ratio of 3:1, whereas the administration of iron with a meal or as a heme iron did not result in a decrease in zinc absorption (IOM 2001). A 56 per cent reduction in iron absorption was observed when zinc was administered in water with a zinc:iron ratio of 5:1; however, when the same ratio was administered in a meal, iron absorption was unaffected (IOM 2001). High concentrations of both tin and copper in the diet may reduce zinc absorption (Valberg *et al.* 1984).

Decreased zinc absorption in association with increased calcium intake was also shown in experiments with swine, leading to a skin condition called parakeratosis (IOM 2001). In human experiments, the impact on zinc absorption depended on the form of calcium administered. Calcium phosphate caused a decrease in zinc absorption while citrate-malate complex did not have a statistically significant effect on zinc absorption. Generally, the consumption of a calcium-rich diet is not expected to significantly affect zinc absorption at normal zinc intake levels. However, the effect of calcium on the absorption of zinc at low zinc intake levels has not been adequately studied (IOM 2001).

Under certain circumstances, zinc and copper may act antagonistically in intestinal absorption. Intake of zinc and copper at normal dietary concentrations does not seem to inhibit absorption. However, if zinc levels are much higher than copper levels, copper absorption decreases (ATSDR 2005; Plum *et al.* 2010). Zinc causes a reduction in copper absorption through the induction of intestinal metallothionein and its subsequent binding to copper.

5.3.1.2 Inhalation Exposure

There are a limited number of quantitative studies pertaining to the inhalation of zinc, although there is evidence that zinc is absorbed via this route, as suggested by increased concentrations in blood and urine from workers exposed to zinc oxide fumes (Hamdi 1969). Inhalation absorption is dependent on both particle size and solubility (ASTDR 2005). One study examined occupational exposure to zinc oxide particulates among galvanization workers in the iron and steel industry (El Safty *et al.* 2008); it identified elevated blood zinc levels in the exposed group compared to the control group. The exposure group also had significantly lower blood copper and calcium levels, suggesting that elevated zinc levels decrease absorption of these metals.

Animal studies provide data on zinc retention in the lungs; however, no studies on absolute absorption were identified. Gordon *et al.* (1992) reported that inhalation exposure to zinc oxide aerosol (3.5-9.1 mg Zn/m^3) for two to three hours resulted in retention values of 19.8, 11.5 and 4.7 per cent for guinea pigs, rats and rabbits, respectively.

5.3.1.3 Dermal Contact

The amount of zinc that passes through the skin will be affected by skin pH, the amount of zinc applied and the vehicle in which zinc is administered (ATSDR 2005). The most water-soluble zinc compounds will be more easily absorbed (ATSDR 2005). Dermal absorption is considered a method of zinc supplementation when oral supplemation is not possible (Keen and Hurley 1977 in OMOE 2011a). HC relies on the OMOECC's assessment for many of its relative absorption factors (RAFs). In the case of zinc, the OMOECC notes that quantitative data are insufficient to determine estimates of the absolute dermal absorption of zinc and many other inorganics (OMOE 2011a). A relative dermal absorption of 1 per cent was selected by the OMOE based on the geometric mean of the midpoint values estimated by US EPA (2004), California Environmental Protection Agency (2000), New York State Department of Environmental Conservation and New York State Deparment of Health (2006) and Massachusetts Deparment of Environmental Protection (1992) (OMOE 2011a).

5.3.2 Distribution

Following oral ingestion of zinc and absorption through the small intestine, distribution occurs via serum, where zinc is bound to proteins such as albumin and α-microglobulin (ATSDR 2005). Upon circulation of zinc in the body, some zinc is biotransformed and retained in storage tissues and the rest is excreted via feces, urine, exhaled air and secretions. Skeletal muscle and bone have been shown to contain over 85 per cent of total body zinc (IOM 2001; 2006). Other body parts known to contain small amounts of zinc are the prostate, liver, gastrointestinal tract, kidney,

skin, lung, brain, heart and pancreas (Plum *et al.* 2010). Regulation of zinc in the body is achieved by a combination of absorption and secretion of endogenous zinc, primarily in the intestine.

5.3.3 Metabolism

Zinc is mostly bound to organic ligands rather than free in solution as a cation (Gordon *et al.* 1992). Absorbed zinc is found in both diffusible and non-diffusible forms in the blood. Approximately two thirds of the diffusible form is exchangeably bound to albumin while the rest is bound to amino acids (mostly histidine and cysteine) (Henkin 1974 in ATSDR 2005). There is equilibrium between the zinc–albumin and zinc–amino acid complex. The non-diffusible form of zinc is tightly bound to alpha2-microglobulin in the plasma (Cousins 1985 in ATSDR 1995).

5.3.4 Elimination

Following inhalation exposure to zinc oxide, workers had elevated urinary zinc levels (Hamdi 1969). Following oral exposure, zinc is primarily eliminated by secretion through the intestine into feces and, to a lesser extent, in urine (ATSDR 2005). As intake increases, fecal and urinary excretions increase (ATSDR 2005). Saliva, secretions, hair loss and sweat contribute minimally to zinc elimination from the body (ATSDR 2005). A low dietary intake of zinc or malnutrition can result in an increase in the proportion of zinc that is eliminated via urinary excretion (ATSDR 2005). No validated physiologically based pharmacokinetic (PBPK) models exist for zinc in humans at this time.

5.3.4.1 Zinc Concentrations in Human Tissues and Bodily Fluids

Zinc was measured in the blood and urine of all participants aged 6–79 years in the Canadian Health Measures Survey (HC 2010a). These data provide reference ranges for blood and urinary zinc levels in the Canadian population. The geometric mean for the concentration of zinc in blood for both men and women was 6.37 mg/L (n=5319, 95 per cent, CI=6.28–6.46). When adjusted for creatinine, the geometric mean was 307.11 µg/g creatinine (n=5479, 95 per cent, CI=296.99–317.57) (HC 2010a). In a study carried out in British Columbia on non-smoking adults aged 30–65, the geometric mean and 95th percentile concentrations of zinc in urine were 285.43 µg/g creatinine and 607.83 µg/g creatinine, respectively (Clark *et al.* 2007). ATSDR (2005) reported zinc concentrations in fingernails, toenails and scalp hair ranging from 94–129 µg/g and concentrations in cadaver tissues levels ranging from 1.5–55 µg/g.

5.4 Mammalian Toxicology

An extensive amount of human health toxicological research has been completed for zinc. Due to the extensive database, all toxicity reference values are based on human studies and thus, laboratory animal studies have generally not been summarized here. Some information on mammalian toxicology can be found in Section 4.4.

5.5 Human Toxicology

5.5.1 Acute Oral Toxicity

Gastrointestinal distress is the primary symptom of acute toxicity of ingested zinc. Symptoms include abdominal pain and cramps, headaches, diarrhea, nausea, vomiting and loss of appetite (IOM 2001). Fosmire (1990) estimated that vomiting and nausea were induced at estimated doses of one to two grams of zinc sulphate salt (225–450 mg of zinc) in adults. This was observed in cases where zinc from galvanized containers had leached into food or drink and the exact doses of the metal were not quantifiable.

5.5.2 Sub-chronic Oral Toxicity

Fischer *et al.* (1984) addressed the effects of zinc supplementation on copper status in a study involving 26 healthy adult male volunteers given either a daily supplement of 50 mg of zinc or a placebo for six weeks. Results showed a decrease in the copper status in the experimental group at the six-week mark and, thus, zinc supplementation was concluded to competitively interact with copper. A lowest observed adverse effect level (LOAEL) of 0.94 mg/kg bw/day was reported (Fischer *et al.* 1984).

Samman and Roberts (1988) investigated the effects of a large dose of zinc (150 mg/d) on plasma lipoproteins and copper status in 26 female and 27 male volunteers aged an average of 27 and 28 years, respectively, in a 12-week double-blind cross-over trial. Subjects were randomly assigned a six-week, three-times-daily 50 mg elemental zinc treatment, followed by a placebo or vice versa. Results demonstrated a reduction in low-density lipoprotein (LDL) levels (-9 per cent) for females but no change in plasma total cholesterol and high-density lipoproteins (HDL) levels in both sexes. Therefore, it was concluded that healthy women consuming 150 mg/day of zinc for six weeks had a decreased risk factor for coronary heart disease due to a shift in cholesterol (LDL-C) from HDL3 to HDL2. However, this result was accompanied by a reduced copper status. Conclusions recommended avoiding prolonged administration of this dose of zinc due to detrimental effects on copper status.

5.5.3 Long-Term/Chronic Oral Toxicity

Bonham *et al.* (2003a; 2003b) evaluated the effects of zinc (supplementation + diet = 30 + 9.29 mg/d) on men in a 22-week study. The study group received 30 mg/d zinc supplements for 14 weeks followed by 8 weeks of copper supplementation to counteract any effects of the zinc supplements on copper status. The control group received placebo supplements throughout the duration of the study. No effect on lipoprotein metabolism, hemostasis or copper status was observed over the three-month period. The authors conclude that an upper limit of 40 mg/d (30 mg/d supplement + 10 mg/d from diet) is safe for the majority of individuals.

A study conducted to evaluate the effects of zinc supplementation on iron, copper and zinc status in adult females found that zinc supplementation poses a risk to iron and copper status (Yadrick *et al.* 1989). The interaction of zinc with iron was studied in 18 adult females using two test

groups: one given a supplement of 50 mg zinc/day, the other given 50 mg of both iron and zinc for 10 weeks. Analysis of blood and saliva samples led the study authors to conclude that zinc supplementation alone represents a risk to iron and copper status whereby zinc supplementation with iron ameliorates the effect on iron, but not on copper.

Nations *et al.* (2008) postulated a relationship between zinc exposure via denture cream and serious neurologic disease. Four patients using two or more tubes of denture cream per week (17,000–34,000 µg Zn/g) were found to have high blood zinc levels and low blood copper levels and suffered from neurologic abnormalities. Detailed medical evaluations failed to identify any other process, aside from the copper deficiency, that may explain the abnormalities. Aside from the denture cream, no other sources of excess zinc ingestion were identified. Patients that stopped using the denture cream and received copper supplementation showed neurologic improvement.

The effects of excessive and deficient zinc intakes on copper metabolism were investigated in a study of 25 healthy postmenopausal women. Groups received diets with either low (1 mg/d) or high (3 mg/d) copper content. Both groups also received 3 mg/d zinc supplementation for the first 90 days and then an additional 50 mg/d for the second 90-day period (Davis *et al.* 2000; Milne *et al.* 2001). Zinc supplementation was found to increase extracellular, but not erythrocyte superoxide dismutase activity, and decrease amyloid precursor protein expression in platelets when combined with a low-copper diet. A positive copper balance resulted only with 3 mg copper + 53 mg zinc. Cholesterol levels were lower with zinc supplementation.

Meta-analysis of clinical trials (n=14,238 subjects from 20 studies with a dose range of 15–160 mg elemental Zn/d) found no effect of zinc supplementation on plasma lipoproteins (Foster *et al.* 2009). However, in healthy individuals, zinc supplementation was associated with a decrease in HDL cholesterol and an increase in total plasma lipids in subjects with conditions that affect zinc homeostasis.

Serum ferritin, hematocrit and erythrocyte Cu, Zn-superoxide dismutase were found to be significantly lower after supplementation (Davis *et al.* 2000; Fischer *et al.* 1984; Milne *et al.* 2001; Yadrick *et al.* 1989). ATSDR (2005), IOM (2001), RIVM (2001) and US EPA (2005) all used decreased erythrocyte superoxide dismutase (ESOD) activity as the critical effect when deriving their toxicological reference values. A decrease in ESOD is not considered an adverse effect in itself but rather a precursor to adverse effects. While ATSDR (2005), IOM (2001), RIVM (2001) and USEPA (2005) all used the inhibition of ESOD as a critical effect for their TRV development, EFSA (2014) and the European Commission (2004) considered otherwise.

The European Commission (2004) considered inhibition of ESOD as an enzymatic change with limited biological significance (Boreiko 2010). Their difference in expert judgment was influenced by research conducted since the other agencies adopted their values (Boreiko 2010), which indicated that there was no association between impaired copper balance and zinc-induced ESOD inhibition. Other authors (Davis *et al.* 2000; Milne *et al.* 2001) reported an increase in deep-tissue ESOD levels as a function of zinc exposure level. Ultimately, the reductions of ESOD was considered to be an adaptive change in response to lower oxidative stress resulting from an increased zinc intake and, thus, not considered as an indicator of copper deficiency (Boreiko 2010). The derivation of the toxicity reference values from numerous international regulatory agencies and organizations is discussed in Section 5.8.

5.5.4 Inhalation Toxicity

Reports on human exposure to zinc by inhalation are mostly limited to occupational settings in which subjects were acutely exposed to zinc oxide fumes during welding or smelting or in military exercises using smoke bombs, which tend to release a caustic zinc chloride smoke (Plum *et al.* 2010). In the case of welding or smelting, the most commonly reported effect is metal fume fever, the symptoms of which, including headache, flu-like symptoms, fever, profuse sweating, chills and weakness (US EPA 2005), appear a few hours after exposure and can persist for up to 48 hours (US EPA 2005). An LOAEL of 5 mg/m^3 was identified for ultrafine zinc oxide particles from steel welding operations (Bodar *et al.* 2005). Zinc chloride fumes from smoke bombs tend to be more injurious to the respiratory system and cases of severe respiratory damage, adult respiratory distress syndrome and mortality have been documented (ATSDR 2005; Plum *et al.* 2010). Bodar *et al.* (2005) identified a LOAEL of 4.8 mg/m^3 for ultrafine zinc chloride.

A higher prevalence of metal fume fever was found in adult males involved in the galvanization process compared to those who were not exposed to increased levels of zinc in the workplace (El Safty *et al.* 2008). There were, however, no long-term effects from zinc exposure based on chest X-rays and the evaluation of ventilator function of both groups.

The US EPA and ATSDR do not provide TRVs for zinc exposure by inhalation. The American Conference of Industrial Hygienists (ACGIH), the Occupational Safety and Health Administration (OSHA) and the National Institute for Occupational Safety and Health provide health-based exposure limits for occupational settings for zinc based on inhalation exposure (ACGIH 1991; OSHA 2012). The levels are based on preventing metal fume fever in the case of zinc oxide and pulmonary toxicity in the case of zinc chloride (ACGIH 1991). This type of exposure is not relevant for contaminated sites and will not be considered in this document.

5.5.5 Dermal Toxicity

Although zinc oxide has been documented to cause skin pox, this was only observed in workers who were involved in packaging large quantities of zinc oxide and chronically covered with a large amount of zinc dust in an observational study in the 1920s (Turner 1921 in ATSDR 2005).

Even though some zinc absorption occurs, dermal exposure to zinc or zinc compounds does not typically result in noticeable toxic effects (ATSDR 2005). Zinc oxide is used in topical applications (e.g., sunscreen and creams). Still, exposure to zinc chloride and other zinc salts can result in severe skin irritancy. RIVM (2001) concluded that as much as 65 per cent of zinc (as zinc oxide) may be absorbed during wound treatment

5.6 Reproductive Effects and Teratogenicity

Zinc is an essential element and thus required for proper human foetal development (ATSDR 2005). Only one study on the reproductive effects of zinc supplementation in pregnant women was identified (Mahomed *et al.* 1989). In this study, 0.3 mg/kg bw/day supplementation of zinc

did not impact changes in maternal body weight gain, blood pressure, postpartum hemorrhage or infection, Apgar scores or birth weight.

However, at much higher doses, reproductive effects of zinc on pregnant sheep were reported by Campbell and Mills (1979). When pregnant sheep were fed 20 mg Zn/kg bw/day during the first 10 days of the gestation period and 10 mg Zn/kg bw/day during the final 10 weeks, 64 per cent of the offspring were non-viable. These authors also observed reduced rates of weight gain and feed consumption by the sheep and lower offspring weights when pregnant sheep were fed zinc.

5.7 Carcinogenicity and Genotoxicity

At the current time, no major health agency considers zinc to have carcinogenic potential. ATSDR (2005) indicates that long-term oral or inhalation exposure to zinc compounds has not been shown to cause an increased incidence of cancer. Similarly, genotoxic effects have not been demonstrated by studies reviewed by the ATSDR, although weak clastogenic effects have been noted (ATSDR 2005).

Based on a retrospective dietary questionnaire, Leitzmann *et al.* (2003) found a relationship between zinc intake and increased prostate cancer risk in a study of men (n=46,974) who consumed more than 100 mg/day of supplemental zinc. Similarly, Gallus *et al.* (2007) reported an association between prostate cancer and zinc intake in a large case-control study of hospitalized males conducted in Italy (n=1,294 cases and 1,451 controls). In this study, the men who consumed the most zinc (based on estimated dose rates from a questionnaire) had elevated rates of prostate cancer compared to men who had lower zinc exposures. Both studies have been criticized by Costello *et al.* (2004) and Costello and Franklin (2007), who concluded that a direct association between high zinc intake and prostate cancer is not supported by the data.

For the purpose of deriving the SQG, zinc was considered to be non-carcinogenic based on the lack of uncontested evidence of any carcinogenic effects in the literature.

5.8 Toxicological Limits

Toxicological reference values are available from numerous international regulatory agencies and organizations. The TRVs developed by international health agencies are summarized in Table 3 with details provided below.

The EFSA (2014) established a UL of 0.37 mg/kg bw/d based on an NOAEL of 50 mg/d from SCF (2003) for multiple indicators of copper status. EFSA applied an uncertainty factor of 2 to the SCF NOAEL.

ATSDR (2005) recommends a minimal risk level (MRL) of 0.3 mg/kg bw/day for chronic oral exposure to zinc. An MRL is defined by the ATSDR (2005) as an estimate of the daily human exposure to a substance that is likely to be without an appreciable risk of adverse effects (non carcinogenic) over a specific duration of exposure. Therefore, it is expected that a chronic oral MRL would be without adverse effects when consumed daily over a long period of time. The MRL is based on an NOAEL of 50 mg/d (0.83 mg/kg bw/day) (extrapolated from Yadrick *et al.*

1989 including only exposure to zinc supplements). TRVs based solely on exposure to supplements are not necessarily appropriate, as this process does not represent total exposure.

RIVM (2001) has derived maximum permissible risk ingestion values (defined as the "amount of a substance that any human individual can be exposed to daily during full lifetime without significant health risk" [RIVM 2001]) and which are equivalent to tolerable daily intakes (TDIs). For zinc, RIVM recommends a TDI of 0.5 mg/kg bw/day. This value is based on an LOAEL of 1 mg/kg bw/d for reduction of ESOD activity based on Yadrick (1989) and cited in ATSDR (2005). A safety factor of 2 was applied to extrapolate an NOAEL from an LOAEL.

The US EPA (2005) recommended a reference dose (RfD) of 0.3 mg/kg bw/day for chronic oral exposure. The US EPA evaluated the available literature and concluded that the inhibition of ESOD might be an indication of mild copper deficiency (Boreiko 2010; Cantilli *et al.* 1994; US EPA 2005). In addition, the US EPA inferred that ESOD inhibition (an enzyme generally regarded as being present in significant excess) may be an indicator of impairment to the antioxidant defense systems (Boreiko 2010; US EPA 2005). This biochemical change was thus judged to represent an LOAEL suitable for the derivation of an RfD through the application of uncertainty factors (Boreiko 2010). The US EPA derived this RfD from an average LOAEL of 0.91 mg/kg bw/day for decreased ESOD activity, which was derived by averaging the results from three principal studies: (i) Yadrick *et al.*'s (1989) LOAEL of 0.99 mg/kg bw/day; (ii) Fischer *et al.*'s (1984) LOAEL of 0.94 mg/kg bw/day; and (iii) Davis *et al.* (2000) and Milne *et al.*'s (2001) LOAEL of 0.81 mg/kg bw/day (two journal articles reporting the same study). An uncertainty factor of 3 to account for human variability was then applied to estimate the RfD of 0.3 mg/kg bw/day.

The RfD derived by the US EPA was considered somewhat controversial since the RfD was lower than the levels of zinc routinely supplied in prenatal vitamin supplements and vitamin formulations for young infants (Boreiko 2010). In addition, it made no allowance for different dietary patterns that might restrict zinc bioavailability. However, it is important to note that the RfD is an estimate with uncertainties that may span an order of magnitude of a daily exposure to the human population (including sensitive subgroups) and is therefore unlikely to be associated with appreciable risk of deleterious effects during a life time (Boreiko 2010).

WHO (1996; 2004) considered many of the same studies as the US EPA and, similarly, considered ESOD inhibition a probable indicator of impaired copper metabolism. WHO selected a total daily intake level of 60 mg/day to represent an LOAEL for effects of modest health significance (Boreiko 2010). WHO stressed that, worldwide, there are different dietary patterns that influence dietary zinc uptake and that an acceptable dietary intake in the United States was likely not applicable in other countries. WHO asserted that the average zinc intake levels for adults (male) should not exceed 45 mg/day (Boreiko 2010; WHO 1996).

The European Commission's Existing Substances Program evaluated the same studies as the US EPA and WHO but concluded that the inhibition of ESOD was an enzymatic change with limited biological significance (ECB 2004). The European Commission recommended an upper limit for zinc of 50 mg/day based on an NOAEL of 50 mg/day in humans leading to inhibition of ESOD and an uncertainty factor of 1 (Boreiko 2010). The key difference between the European Commission's evaluation and those of the US EPA and WHO is the use of an NOAEL instead of an LOAEL as the point of departure for ESOD inhibition in humans (Boreiko 2010).

TDIs or RfDs for essential trace elements (ETEs) may be overly conservative when compared to dietary reference intakes (DRIs) established by the Food and Nutrition Board of the IOM (2000; 2001). In some instances, when the same data sets are used to develop both TDIs and DRIs, uncertainty in the data results in these values being set at the lower and higher ends of their ranges, respectively (Goldhaber 2003). DRIs consider bioavailability as well as all nutrient and dietary interactions (IOM 2000, 2001; Mertz 1995; WHO 2002) and are normally developed for specific age and gender groups and physiological states for almost all population groups. Hence, different values can protect sub-population groups at risk without being over-protective for the rest of the general population (Mertz 1998; Munro 1999). Tolerable upper intake levels (UL) consider risks from both nutrient deficiencies and toxicity and the variability between individuals (WHO 2002). For ETEs, there is a safe range of intakes between deficiency and toxicity that is generally represented by a U-shaped "dose-response" or risk probability curve (Abernathy 1999; Becking 1998; WHO 1996). ULs are not specific data points from any particular dose-response study, but are derived using well-established principles of risk assessment (WHO 2002).

The use of large uncertainty factors (UFs) may conceivably lead to values associated with nutritional deficiencies. UFs used to establish ULs are usually much less than those used to calculate TRVs because of the availability of reliable human data (Becking 1998; Dourson *et al.* 2001; Munro 1999). ULs are not recommended levels of intake (i.e., recommended daily allowance). Direct toxicity appears to be absent at doses above the recommended daily allowance but less than or equal to the UL. Hence, overestimating the toxicity of ETEs at contaminated sites may become costly when ETEs are drivers for site management, including remediation.

The Contaminated Sites Division has adopted an approach for establishing TRVs for essential trace elements that reflects the understanding of the benefits and risks posed by these substances. HC (2009) selected the IOM (2001) UL as its zinc TDI or RfD for ingestion exposure for the purpose of derivation of soil quality guidelines. Adjustments for relative bioavailability may be necessary when considering exposure *via* soil and/or water ingestion.

Table 3: Summary of Zinc TRVs

Agency	Classification	Estimated TRV[i]	Evaluation	Notes
EFSA (2014)	DRI	0.37 mg/kg bw/d	Sub-chronic oral exposure to Zn supplements	NOAEL of 50 mg/kg based the absence of any adverse effect on a wide range of relevant indicators of copper status in controlled metabolic studies in adults including pregnant and lactating women (SCF 2003).
ATSDR (2005)	MRL	0.3 mg/kg bw/day	Intermediate and chronic oral exposure	MRL based on NOAEL = 0.83 mg/kg bw/day and an uncertainty factor of 3 to account for human variability. Principal study cited is Yadrick *et al.* (1989) and supplemented by Milne *et al.* (2001); Davis *et al.* (2000); Black *et al.* (1988); Fischer *et al.* (1984); Freeland-Graves *et al.* (1982) and Prasad *et al.* (1978).
RIVM (2001)	Maximum permissible risk (equivalent to TDI)	0.5 mg/kg bw/day	Chronic oral exposure	Based on LOAEL = 1.0 mg/kg bw /day and a safety factor of 2 to convert from an LOAEL to an NOAEL. ATSDR (1995) is cited as source of LOAEL.
IOM (2001)	Tolerable UL	4 mg/day (0.49 mg/kg bw/day)[a] 40 mg/day (0.57 mg/kg bw/day)[a]	Chronic oral exposure, infants 0–6 months Chronic oral exposure to adults	UL based on NOAEL = 4.5 mg/day for infants and application of uncertainty factor of 1. UL based on LOAEL = 60 mg/day for adults and then application of uncertainty factor of 1.5 to account for human variability and use of an LOAEL. NOAEL based on Walravens and Hambidge (1976); LOAEL based on Yadrick *et al.* (1989) and supported by Fischer *et al.* (1984).
US EPA (2005)	RfD	0.3 mg/kg bw/day	Chronic oral exposure	RfD based on average LOAEL = 0.91 mg/kg bw/day and an uncertainty factor of 3 to account for variability in susceptibility in human populations. Principal studies were cited as Yadrick *et al.* (1989); Milne *et al.* (2001); Davis *et al.* (2000); Fischer *et al.* (1984).
Joint Expert Committee on Food Additives (JECFA) (1982)	Provisional maximum TDI	1 mg/kg bw/day	Chronic oral exposure	Specific details on the derivation of this value were not provided.

Notes
[i]The estimated TRV in mg/kg bw/day was calculated from the IOM (2001) UL using CCME (2006) body weights of 8.2 kg for an infant and 70.7 kg for an adult, respectively.

IOM (2001) provided a range of tolerable ULs of 4 mg/day for infants to 40 mg/day for adults. The UL for infants was based on an NOAEL of 4.5 mg/day reported by Walravens and Hambidge (1976), where no adverse health effects were found in infants receiving total zinc in milk at 5.8 mg/L (1.8 mg + 4 mg supplement) at an estimated 0.78 L/day. An uncertainty factor of 1 was applied, due to a large number of infants in the study and the value was rounded down to 4 mg/day for infants aged 0 through 6 months.

For adults, the IOM (2001) based the UL on an LOAEL of 60 mg/day of zinc (50 mg/day supplemental zinc plus 10 mg/day estimated dietary intake) that resulted in a reduced copper status in healthy women (Yadrick *et al.* 1989). This LOAEL is also supported by other investigations showing altered copper status after zinc supplementation (Fischer *et al.* 1984). An uncertainty factor of 1.5 was applied to account for inter-individual variability in sensitivity and for the extrapolation from an LOAEL to an NOAEL. This calculation resulted in the derivation of a tolerable UL for zinc from food, water and supplements for adults of 40 mg/day (IOM 2001).

Table 4: IOM- and HC-adjusted Zinc ULs for Various Age and Gender Groups

IOM (2001)					HC adjustments			
Receptor	Age group	Body weight (kg)	Tolerable UL		Receptor	Age group	Body weight (kg)	Derived values (mg/kg-d)
			mg/day[1]	mg/kg-d[2]				
Infant	0–6 mo	7	4	0.57	Infant	0–6 mo	8.2	0.49
	7–12 mo	9	5	0.56	Toddler	7 mo–4 yrs	16.5	0.48
Children	1–3 yrs	13	7	0.54				
	4 yrs	22	12	0.55				
	5–8 yrs				Children	5–11 yrs	32.9	0.51
	9–11 yrs	40	23	0.58				
	12v13 yrs				Adolescent	12–19 yrs	59.7	0.54
Adolescent	14–18 yrs	64 (M)	34	0.53				
		57 (F)		0.60				
Adult	19 yrs	76 (M)	40	0.53				
		61(F)		0.66				
	≥ 20 yrs	76 (M)	40	0.53 (M)	Adult	≥ 20 yrs	70.7	0.57
		61(F)		0.66 (F)				
Pregnant and lactating women	14–18 yrs	–	34	–	–	–	–	–
	19–50 yrs		40					

1: IOM (2001)
2: ULs in mg/day divided by body weights (in kg) as reported in IOM (2001)
F: Female, M: Male

The IOM (2001) tolerable UL was adjusted to be consistent with body weights from CCME (2006) (Table 4). For toddlers (7 mo–4 yr) weighing 16.5 kg, the tolerable UL is estimated at 0.48 mg/kg bw/day (i.e., [5 mg/day x 6 months + 7 mg/day x 36 months + 12 mg/d x 12 months]/54 months/16.5 kg = 0.48 mg/kg bw/day). For a 70.7 kg adult, the tolerable UL is estimated at 0.57 mg/kg bw/day (i.e., 40 mg/day/70.7 kg = 0.57 mg/kg bw/day). These values were used as the TDI for SQG_{DH} development.

The Joint FAO/WHO Expert Committee on Food Additives (JECFA) evaluates the safety of naturally occurring substances. JECFA (1982) reports a provisional maximum TDI for zinc of 1.0 mg/kg bw/day. Specific details on the derivation of this value were not provided.

No major health agency provides a TDI or tolerable concentration for zinc (as particulates) via the inhalation route. Information from occupational studies was not appropriate for derivation of an SQG. Consequently, the oral TDIs described above were used for evaluation of exposures to zinc from all routes of exposure including inhalation.

5.8.1 Overall Conclusions

For the purposes of SQG_{HH} development, zinc has been evaluated as a non-carcinogen. All TRVs from major health agencies were based on the oral route of exposure since dietary intake is the primary route of exposure in humans. For the current assessment, the IOM (2001) tolerable ULs were adopted as the TDIs for zinc after averaging the values over body weight in kilograms. The following TDIs were used in development of the SQG_{DH}:

TDI for toddlers: 0.48 mg/kg bw/day

TDI for adults: 0.57 mg/kg bw/day

6.0 DERIVATION OF ENVIRONMENTAL SOIL QUALITY GUIDELINES

The derivation of environmental soil quality guidelines for zinc is outlined in the following sections for four land uses: agricultural, residential/parkland, commercial and industrial. These guidelines have been developed according to the protocol described by CCME (2006). The information presented in this chapter adds to the environmental data presented in original 1999 zinc soil quality guidelines Scientific Supporting Document developed by Environment Canada, for CCME (EC 1999a).

The environmental soil quality guidelines for zinc are derived using the available toxicological data to determine the threshold level of effects for key ecological receptors. Exposure from direct soil contact is the primary derivation procedure used for calculating environmental quality guidelines for residential/parkland, commercial and industrial land uses. Exposure from direct soil contact as well as soil and food ingestion are considered in calculating guidelines for agricultural land use, with the lower of the two values generated from these derivation procedures being recommended as the environmental soil quality guideline for this land use. In addition to these primary derivation procedures, check mechanisms are used to consider

important direct and indirect soil exposure pathways, such as the nutrient and energy cycling check.

All data selected for use in the following derivations have been screened for ecological relevance. Note that *E. fetida* is known to inhabit Canadian soils. The selected data for plants and invertebrates used in the derivation of the guidelines for soil contact are presented in Appendix XIV while Appendix XIII presents selected microbial studies used in the nutrient and energy cycling check. The SQG_I for soil and food ingestion was derived using the selected data shown in Appendix XV. Studies were excluded from use because of one or more of the following reasons:

1. Soil pH was not recorded.
2. Soil pH was below 4 (since this is considered outside the normal pH range of most soils in Canada).
3. No indication of soil texture was provided.
4. Inappropriate statistical analysis was used.
5. Test soil was amended with sewage sludge or a mixture of toxicants.
6. Test was not conducted using soil or artificial soil.
7. Test did not use controls.

LOEC and EC_x data used in the following derivations were considered to be statistically significant according to the study from which the data were taken. Alternatively, for endpoints reported simply as EC_x, guideline developers calculated the percentage of adverse effects, as compared to controls, from data presented by the study author.

6.1 Soil Quality Guidelines for Agricultural and Residential/Parkland Land Uses

6.1.1 Soil Quality Guideline for Soil Contact

The derivation of the soil quality guideline for soil contact (SQG_{SC}) is based on toxicological data for vascular plants and soil invertebrates. The toxicological data for plants and invertebrates used in guideline derivation are presented in Appendix XIV. A total of 17 plant studies covering 19 species and 98 endpoints, and 13 invertebrate studies covering 9 species [note: nematodes not reported to species level] and 68 endpoints were acceptable for use (i.e., toxicity studies that were classified as “selected”).

The preferred Weight of Evidence derivation method using an EC_{25}/IC_{25} distribution could not be performed because the invertebrate dataset lacks these endpoints. The plant and invertebrate data were combined (166 endpoints; Appendix XIV) and a guideline was derived using the Weight of Evidence Method using an Effects/No Effects data distribution (Section 7.5.5.2 of CCME 2006). The endpoints were ranked and the 25th percentile of the estimated species sensitivity distribution ($ESSD_{25}$) was used as the basis for soil contact guidelines for the agricultural and residential/parkland land uses (Figure 1).

The threshold effects concentration (TEC) was calculated as follows:

$$\text{TEC} = \frac{\text{ESSD}_{25}}{\text{UF}}$$

where,

TEC = threshold effects concentration (mg/kg) – i.e. guideline value
$ESSD_{25}$ = estimated species sensitivity distribution – 25th percentile of the distribution (mg/kg)
UF = uncertainty factor (if needed); no uncertainty factor was applied

The 25th percentile of the ESSD resulted in a rank of 41.75. The 41st and 42nd ranked endpoints were both 250 mg/kg.

Thus,

The TEC = 250 mg Zn/kg soil.

The SQG_{SC} for Agricultural and Residential/Parkland land use is set at the TEC.

6.1.2 Nutrient and Energy Cycling Check

The soil quality guideline for the protection of nutrient and energy cycling (SQG_{NEC}) was calculated using the selected microbial processes data presented in Appendix XIII. Nitrification and nitrogen fixation data are considered to be primary data, whereas nitrogen mineralization, denitrification and carbon cycling data are considered secondary data. LOEC data, as reported by the author, are used directly while EC_x data producing >15 and <40 per cent effects in primary data (i.e., EC_{15} to EC_{40}) and >15 and <25 per cent effects in secondary data (i.e., EC_{15} to EC_{25}) are interpreted as LOEC values. Insufficient primary data were available for the calculation, so the primary and secondary data were combined and the check was carried out using a modified LOEC method whereby the geometric mean of available LOECs was calculated as the nutrient and energy cycling check.

The soil quality guideline for the protection of nutrient and energy cycling (SQG_{NEC}) is calculated as follows:

$$SQG_{NEC} = (LOEC_1 \times LOEC_2 \times LOEC_3 \times \ldots LOEC_n)^{1/n}$$

where,
SQG_{NEC} = nutrient and energy cycling check(mg/kg^{-1} soil)
LOEC = lowest observed effects concentration (mg/kg^{-1} soil)
n = number of available LOECs

Thus,
NEC $= (327\times327\times327\times1074\times10\times10\times100\times327\times327\times327\times327\times1074\times5000\times620)^{1/14}$
= 275 mg Zn/kg soil

Rounded to two significant figures, the SQG_{NEC} for Agricultural and Residential/Parkland land use is 280 mg Zn/kg soil.

6.1.3 Soil Quality Guideline for Soil and Food Ingestion

The soil quality guideline for soil and food ingestion (SQG_I) applies to agricultural land use, and residential/parkland if substances are considered to bioaccumulate and/or biomagnify. The SQG_I involves determining the grazing or foraging species that are most threatened by contaminated soil and food ingestion. The daily threshold effect dose (DTED) and the soil and food ingestion rates are determined for the species. This information, along with the mean body weight of the species and the bioconcentration factor for the contaminant, are used to calculate the SQG_I that will prevent grazing animals from being exposed to more than 75 per cent of the DTED resulting from the ingestion of soil and plants.

Calculation of the SQG_I to protect primary consumers is based on the LOAEL taken from the selected mammalian and avian toxicological data listed in Appendix XV. The lowest observed adverse effects level, indicating the species most threatened, was 10 mg Zn/kg bw/day for the final 10 weeks of an experiment with sheep resulting in a significant reduction in the number of viable offspring produced (Campbell and Mills 1979).

The LOAEL is used to calculate the DTED according to the equation:

$$DTED = {lowest\ LOAEL}/{UF}$$

where,

DTED = daily threshold effects dose (mg/kg bw-day)
LOAEL = lowest observed adverse effects dose (mg/kg bw-day)
UF = uncertainty factor; no uncertainty factor was applied.

Thus,

DTED = 10 (mg/kg bw-day)

An animal may be exposed to a contaminant by more than one route. Total exposure comes from a combination of contaminated food, direct soil ingestion, dermal contact, contaminated drinking water and inhalation of air and dust. Exposure from all of these routes should not exceed the DTED. Assuming that drinking water, dermal contact and inhalation account for 25 per cent of the total exposure (CCME 2006), the remaining 75 per cent of exposure is attributed to the ingestion of food and soil. It follows then, that exposure from soil and food ingestion should not exceed 75 per cent of the DTED:

exposure from direct soil ingestion + exposure from food ingestion = 0.75 × DTED

6.1.3.1. Exposure from Direct Soil Ingestion

To estimate the exposure of an animal from direct soil ingestion, the rate of soil ingestion must be calculated. The ingestion rate of soil and forage together is referred to as the dry matter intake

rate (DMIR). To estimate the rate of soil ingested directly, the percentage of the DMIR attributed to soil ingestion must be isolated. In most soil-based exposure studies, the proportion of soil ingested (PSI) is reported with the DMIR. The animal's soil ingestion rate is calculated as a proportion of the DMIR according to the equation:

$$SIR = DMIR \times PSI$$

where,

SIR = the soil ingestion rate (kg dw soil/day)
DMIR = geometric mean of available dry matter intake rates (kg dw/day), which was determined to be 1.89 kg/day (Campbell and Mills 1979).
PSI = geometric mean of available soil ingestion proportions reported with DMIR. As no information is available on the PSI for the species used, a default value of 0.083 (McMurter 1993) was used for the above equation.

Thus,

$$SIR = 1.89 \times 0.083 = 0.16 \ kg\ dw\ soil/day$$

The SIR can then be combined with the bioavailability factor (BF), body weight (BW) and a concentration of the contaminant in the soil (SQG_I) to represent the exposure from soil ingestion. The soil concentration at this point is unknown, but it should not provide for greater than 75 per cent of the DTED when combined with the exposure calculated for food ingestion:

$$exposure\ from\ soil\ ingestion = SIR \times BF \times SQG_I/BW$$

where,

SIR = soil ingestion rate (kg dw soil/day)
BF = bioavailability factor; due to a lack of specific information on the bioavailability of zinc from ingested soil for livestock and terrestrial wildlife, a BF of 1 is assumed (CCME 2006).
SQG_I = concentration of the contaminant in soil that will not result in greater than 75 per cent DTED (mg Zn/kg soil)
BW = mean body weight (kg); the mean body weight of sheep was determined to be 80.0 kg (Campbell and Mills 1979).

6.1.3.2. Exposure from Food Ingestion

Similar to SIR, the food ingestion rate (FIR) for livestock and wildlife is expressed as a portion of DMIR. The FIR is the remaining proportion of the DMIR minus soil ingestion rate. The FIR is calculated as:

$$FIR = DMIR - SIR$$

where,

FIR = food ingestion rate (kg dw food/day)
DMIR = geometric mean of dry matter intake rates (kg dw/day)
SIR = soil ingestion rate (kg dw soil/day)

Thus,

$FIR = 1.89 - 0.16 = 1.73\ kg\ dw\ food/day$

The FIR can then be combined with the bioconcentration factor (BCF), BW and the SQG_I to express the exposure from food ingestion:

$$exposure\ from\ food\ ingestion = FIR \times BCF \times SQG_I/BW$$

where,

FIR = food ingestion rate (kg dw food/day)

BCF = bioconcentration factor; (calculated as 0.26 from the geometric mean of data obtained from consulted studies, calculated according to CCME [2006]; see Appendix XVI and XVII)

SQG_I = concentration of the contaminant in soil that will not result in greater than 75 per cent DTED (mg Zn/kg soil)

BW = mean body weight (kg); the mean body weight of sheep was determined to be 80.0 kg (Campbell and Mills 1979).

6.1.3.3.Exposure from Direct Soil Ingestion and Food Ingestion

The equations for exposure from soil ingestion and exposure from food ingestion can be combined and rearranged to solve for the SQG_I:

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

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

$$SQG_I = (0.75 \times 10 \times 80.0)/[(0.16 \times 1) + (1.73 \times 0.26)]$$

$$SQG_I = 984\ mg\ Zn/kg\ soil$$

Therefore, rounding off, the soil quality guideline for exposure through ingestion of soil and food is 980 mg Zn/kg soil for agricultural land use.

6.2 Soil Quality Guidelines for Commercial and Industrial Land Uses

6.2.1 Soil Quality Guidelines for Soil Contact

For commercial and industrial land uses, a lower level of protection and a smaller array of receptors are envisioned than for agricultural and residential/parkland land uses. Therefore, the SQG_{SC} for both commercial and industrial land uses is derived at the level of an effects concentration—low (ECL), rather than a TEC.

The derivation of the SQG_{SC} for these land uses is also based on toxicological data for vascular plants and soil invertebrates presented in Appendix XIV. The ECL was calculated using the Weight of Evidence Method with an Effects/No Effects data distribution. The endpoints were

ranked and the 50th percentile of the estimated species sensitivity distribution ($ESSD_{50}$) was used as the basis for soil contact guidelines for the commercial and industrial land uses (Figure 1).

The ECL was calculated as follows:

$$ECL = ESSD_{50}$$

where,

ECL = effects concentration—low (mg/kg) – i.e., guideline value
$ESSD_{50}$ = estimated species sensitivity distribution – 50th percentile of the distribution (mg/kg)

The 50th percentile of the ESSD resulted in a rank of 83.5. The 83rd and 84th ranked data points are 444 mg/kg and 450 mg/kg, respectively. A value was interpolated for rank 83.5 (50th percentile) as follows;

$ESSD_{50}$= rank 83 + 0.5 × (rank 84– rank 83)

= 444 mg/kg + 0.5 × (450 mg/kg – 444 mg/kg)

= 447 mg/kg

Thus,

the ECL = 447 mg Zn/kg soil

The SQG_{SC} for agricultural and parkland/residential land use is set at the ECL rounded off to 2 significant figures (i.e., 450 mg Zn/kg soil).

6.2.2 Nutrient and Energy Cycling Check

The NEC was calculated using the selected microbial processes data presented in Appendix XIII. Nitrification and nitrogen fixation data are considered to be primary data, whereas nitrogen mineralization, denitrification, and carbon cycling data are considered secondary data. LOEC data, as reported by the author, are used directly while effects concentration (EC) data producing >15 and <50 per cent effects in primary data (i.e., EC_{15} to EC_{50}) and >15 and <35 per cent effects in secondary data (i.e. EC_{15} to EC_{35}) are interpreted as LOEC values. Insufficient primary data were available for the calculation, so the primary and secondary data were combined and the check was carried out using a modified LOEC method whereby the geometric mean of available LOECs was calculated as the nutrient and energy cycling check.

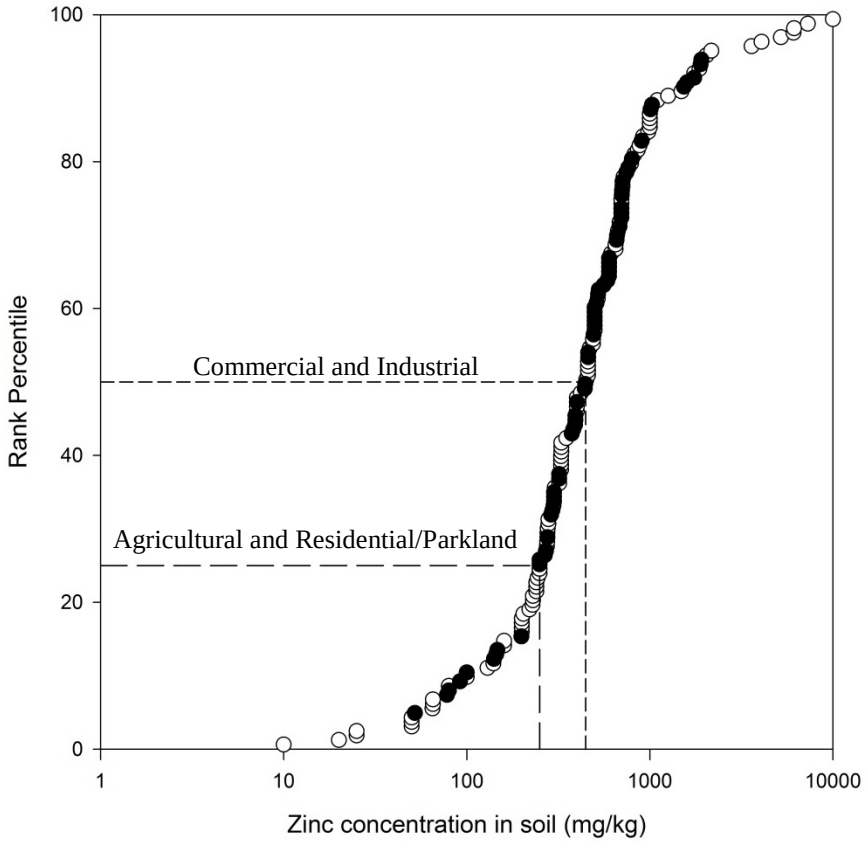

Figure 1: Rank Probability Plot of Zinc Plant and Invertebrate Bioassay Data
Plant (○) open circle, invertebrate (●) closed circle. (Agricultural and Residential/Parkland derivation in Section 6.1.1., Commercial and Industrial land uses in Section 6.2.1.)

The soil quality guideline for the protection of nutrient and energy cycling check (SQG_{NEC}) is calculated as follows:

$$SQG_{NEC} = (LOEC_1 \text{ x } LOEC_2 \text{ x } LOEC_3 \text{ x} \ldots LOEC_n)^{1/n}$$

where,

SQG_{NEC} = nutrient and energy cycling checkcheck (mg Zn/kg soil)
LOEC = lowest observed effects concentration (mg Zn/kg soil)
n = number of available LOECs

Thus,

$$\text{NECNEC} = (327 \times 327 \times 327 \times 1074 \times 1074 \times 10 \times 10 \times 100 \times 327 \times 327 \times 327 \times 327 \times 1074 \times 3000 \times 3000 \times 33 \times 3270 \times 5000 \times 620 \times 240 \times 440 \times 200 \times 559 \times 1908 \times 687 \times 210 \times 850\ )^{1/27}$$

= 412 mg Zn/kg

Rounded off to two significant figures, the SQG_{NEC} for commercial and industrial land use is 410 mg Zn/kg soil.

6.2.3 Environmental Soil Quality Guideline for Off-site Migration

When deriving soil quality guidelines for commercial and industrial sites, exposure scenarios only consider on-site exposure. However, transfers of contaminated soil from one property to another are possible by environmental routes such as wind and water erosion (CCME 2006).

The environmental soil quality guideline for off-site migration (SQG_{OM-E}) refers to the concentration in soil eroded from a commercial or industrial site that will raise the contaminant concentration in an adjacent, more sensitive land (e.g., agricultural property) within a specific time frame. The purpose or the SQG_{OM-E} is to establish commercial or industrial soil guidelines that will not result in unacceptable adverse effects (i.e., not to exceed agricultural guideline) to more sensitive land uses due to contaminant migration over a specified time period. The SQG_{OM-E} was derived as follows:

$$SQG_{OM-E} = 14.3 \times SQG_A - 13.3 \times BSC$$

where,

SQG_{OM-E} = environmental soil quality guideline for off-site migration (mg/kg);

SQG_A = environmental soil quality guideline (SQG_E) for agricultural land use (250 mg/kg; see Table 5);

BSC = background soil concentration (48.1 mg/kg; Grunsky 2010a);

The environmental soil quality guideline for off-site migration (SQG_{OM-E}) is 2,935 mg/kg. Rounded to two significant figures, the SQG_{OM-E} is 2,900 mg/kg.

6.3 Final Environmental Soil Quality Guidelines

The final environmental soil quality guidelines for zinc for agricultural, residential/parkland, commercial, and industrial land uses are presented in Table 5. The guidelines are applicable to soils within the pH range of 4.0 to 8.3, as the toxicological studies upon which these guidelines are based were conducted within this pH range.

6.3.1 Agricultural Land Use

The lower value from the three procedures (SQG_{SC}, SQG_I and SQG_{NECC}) is selected as the final environmental soil quality guideline for agricultural land. The lower of the three procedures is the SQG_{SC}. Therefore, the final SQG_E for agricultural land use is 250 mg Zn/kg dry soil.

6.3.2 Residential/Parkland Land Use

For contaminants that do not bioaccumulate and or biomagnify, the lowest of the SQG_{SC} and SQG_{NEC} is used as the SQG_E. The lower of the two procedures is the SQG_{SC}. Therefore, the final SQG_E for residential/parkland land use is 250 mg Zn/kg dry soil.

6.3.3 Commercial and Industrial Land Use

The SQG_{SC} for commercial and industrial land use is 450 mg Zn/kg dry soil. This value is higher than the NECC value of 410 mg/kg soil and is thus not protective of microbial processes. Therefore, the final SQG_E for commercial and industrial land use is 410 mg Zn/kg dry soil, based on the NECC. The SQG_E was not modified by the SQG_{OM-E}.

7.0 DERIVATION OF HUMAN HEALTH SOIL QUALITY GUIDELINES

7.1 Protocol

Human health soil quality guidelines provide concentrations of substances in soil at or below which no appreciable risk to human health is expected. In order to derive a quantitative guideline, it is necessary to define one or more scenarios by which exposure will occur. The majority (88.9 per cent) of Canadians obtain drinking water from a treated water distribution system. Some Canadians obtain water from private wells (10.5 per cent) and trucked water (0.6 per cent) (EC 2011). Therefore, the most likely route of exposure to substances in soil is expected to be direct contact with soil for the majority of Canadians. Receptor characteristics and estimated daily intakes used to calculate the human health SQGs are summarized in Appendix IV and VII and are discussed in the relevant sections of this document.

Human health Canadian soil quality guidelines are defined for agricultural, residential/parkland, commercial and industrial land uses for soil ingestion, dermal contact with soil and inhalation of soil particulates according to the *Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 2006). Inhalation of zinc from fugitive dust from soils is not considered to contribute significantly to total exposure or to cause significant route-specific toxic effects similar to occupational exposure settings such as welding where “metal fume fever” has been reported. As such, inhalation exposure does not warrant a route-specific TRV. Therefore, a single guideline for combined ingestion, inhalation and dermal contact exposures was developed for each land use category.

Zinc is not considered to be carcinogenic to humans via any route of exposure (see Section 5.7). For threshold toxicants, two key factors are considered in the setting of SQGs in Canada. First, it is recognized that, exclusive of hazardous waste sites or other point sources of pollution, everyone is exposed to a “background” level of contamination. For zinc (as total), this background exposure arises primarily from foods. In setting SQGs for zinc (as total), the background EDI was subtracted from the TDI before guidelines were derived using the approach

outlined by CCME (2006). In addition to using the CCME (2006) approach, an alternative approach (Appendix XVIII) was also used to set SQGs for zinc (as total) to address issues when the EDI approaches or exceeds the TDI.

Second, a portion of the residual provisional tolerable daily intake (TDI−EDI) must be attributed to each medium in a manner such that total simultaneous exposure at the guideline levels for all media will not result in exposure that exceeds the TDI. As recommended by CCME (2006), 20 per cent of the residual TDI for threshold (non-carcinogenic) toxicants was apportioned to each environmental medium, namely air, water, soil, food and consumer products.

In addition to the direct contact SQGs, the CCME (2006) protocol ensures the protection of groundwater used as a drinking water source and includes two check values: 1) consumption of produce, meat and milk and 2) off-site migration of contaminated soil. In the case of the zinc SQG, an off-site migration check was performed, but the check mechanism for consumption of produce, meat and milk was not required because zinc is not a substance that bioaccumulates. No guideline for protection of groundwater used as a source of raw water as drinking water was calculated for zinc due to constraints on the mathematical model when applied to inorganic compounds (CCME 2006).

7.2 Estimated Daily Intake for Canadians

EDIs for the Canadian population have been calculated on the basis of the environmental concentration of zinc in uncontaminated environmental media (see Section 2.5). In general, the EDI is an estimate of the typical total concurrent background exposure from all known or suspected sources via a multi-media exposure assessment for the average Canadian. It does not include exposures that may occur from a contaminated site or activities that may result in increased exposure of substances that are not considered background exposure.

The EDI calculation is illustrated in the following equation (CCME 2006):

$$EDI = \sum_{i=1}^{n} ED_i$$

The EDIs are expressed in units of mg/kg bw/day and they are intended to represent the average exposure that the Canadian population may receive from zinc. The general population was subdivided into five age classes: infants (birth to 6 months), toddlers (7 months to 4 years), school age children (5 to 11 years), teenagers (12 to 19 years) and adults (20 years and older). The following media were considered in calculating the EDI: ambient air, indoor air, indoor dust, soil, drinking water, food and breast milk. Consumer products were not included in the EDI estimation because there are limited data in this area. The equation below illustrates the media and pathway-specific EDI calculation (CCME 2006).

$$ED_i = \frac{C \times CR \times BF \times EF}{BW}$$

where,

ED_i = exposure dose from pathway *i* (mg/kg-day)
C = contaminant concentration in medium (e.g., mg/L)
CR = media specific contact rate (e.g., L/day)
BF = bioavailability factor (unitless)
EF = exposure factor, which is the product of the exposure frequency (events/year) and exposure duration (years/lifetime) and is unitless
BW = body weight (kg)

Concentrations of zinc in environmental media were obtained from governmental databases, scientific literature and grey literature as summarized in Section 2.5 and presented in Appendix I. Data were selected for EDI determination based on a scoring system to ensure a minimum data quality (HC 2011a), except for soil, ambient air and foods data, which were provided directly from Canadian government databases and treated as probability density functions (PDFs) using Crystal Ball® (version 11.1.1.0.00).

Receptor characteristics and intake rates for each age class were treated as PDFs, as described in HC (2011a). PDFs were assumed to be lognormal, except for human breast milk intake and time spent outdoors, for which a triangular distribution was used due to limited data availability. Dietary intakes were derived deterministically.

PDFs were generated for concentrations in environmental media, receptor characteristics and intake rates. These were used to generate EDI distributions by age group for each media and a combined total EDI from all media and exposure routes using Crystal Ball® (10,000 iterations). Therefore, the total EDI is generated separately and is not simply the sum of the separate EDIs generated for each media per age group. Appendices IV, V and VI list the receptor characteristics used to develop the EDIs including: body weights and intake rates of air, drinking water, soil and dust for each specified age group of the population. Appendix VII summarizes the daily intake estimates for zinc via all media for five age groups of the Canadian general population (HC 2011a).

The total zinc EDIs for adults, teenagers, children and toddlers are 0.178, 0.251, 0.387 and 0.545 mg/kg bw/day (median values), respectively. Depending on whether infants are exclusively formula fed, exclusively breast fed or fed a mixture of breastmilk, infant formula and table food, the EDI for infants is 0.253 mg/kg bw/day (exclusively breastfed) or 0.712 mg/kg bw/day (non-breastfed). For the purpose of soil quality guidelines derivation, the EDI for toddlers and adults were used. Dietary intake was the primary contributor to exposure, accounting for 98–100 per cent of the total EDI (HC 2011a).

Certain Canadian subpopulations may be exposed to higher levels of zinc than the calculated EDIs. High levels of zinc may occur in drinking water due to plumbing and usage patterns. Consumption of such waters would be the most likely route for higher-than-average Canadian exposure to zinc. Consumption of food grown in soils containing high levels of zinc could also possibly increase exposure above the levels calculated in the present exposure analysis. In addition, people living near industrial areas associated with zinc emissions could be exposed to higher concentrations via inhalation of ambient air. Due to insufficient data, it is not possible, at this time, to perform an exposure assessment for those groups. However, current analysis does

suggest that, compared to food consumption, the direct contact pathways for soil (incidential ingestion, inhalation and dermal contact) are small contributors to total zinc exposure.

7.3 Exposure Limits for Human Receptors

As stated in Section 5.8.1, the IOM (2001) tolerable ULs were adopted as the TDIs for zinc after averaging the values over body weight in kilograms. The toddler reference dose of 0.48 mg/kg bw/day and the adult reference dose of 0.57 mg/kg bw/day were considered to be appropriate as the TDIs for calculating SQGs for zinc. Insufficient data were available for derivation of exposure limits for inhalation or dermal exposures, thus the oral TDIs were used to estimate toxicity from inhalation and dermal routes of exposure.

7.4 Relative Absorption Factors

RAFs may be applied when the critical toxicological study has used a different route than that under investigation, in order to account for the difference in absorption of the contaminant by the body from two different media or when TRVs from one route of exposure are applied to another route of exposure.

To develop a soil quality guideline, it is necessary to estimate the relative (to the oral route) dermal absorption factor (which is the basis of the TRV for zinc). This can be done according to the following formula:

$$Relative\ dermal\ absorption\ factor = \frac{Absolute\ dermal\ absorption\ rate}{Absolute\ oral\ absorption\ rate\ in\ TRV\ study}$$

The IOM (2001) tolerable ULs were adopted as the oral TDIs for zinc. These were derived primarily from the study by Yadrick *et al.* (1989), where female participants were given oral zinc supplements. As mentioned in the toxicokinetics section, (Section 5.3), ATSDR (2005) indicated that zinc absorption in humans ranged from 8 to 81 per cent following short-term exposure to zinc supplements.

Insufficient information is available to determine the bioavailability of zinc in soils. Therefore, a relative absorption factor of 100 per cent was selected for exposure via ingestion. Similarly, no data were available for bioavailability via inhalation and a factor of 100 per cent was selected.

OMOE (2011a) recommends a dermal relative absorption factor of 0.1 as a generic default assumption for metals, which is not specific to zinc. Quantitative data are insufficient to estimate the dermal absorption factor for zinc; therefore, the OMOE selected the default RAF of 0.1 using an order-of-magnitude approach (default absolute dermal absorption of 1 per cent is approximately an order of magnitude lower than the estimated absolute oral absorption). For the purposes of this assessment, an RAF of 0.1 (10 per cent) was used for dermal exposure. This approach is deemed reasonable since it errs on the safe side by using the lowest oral absorption observed.

7.5 Human Health Soil Quality Guidelines

7.5.1 Agricultural and Residential/Parkland Land Uses

For purposes of determining an agricultural and residential/parkland soil quality guideline for a threshold substance, it has been assumed that the receptor with the greatest exposure per unit mass is the most sensitive. Based on the general characteristics attributed to the Canadian population (HC 1994; Richardson 1997), this is a toddler aged >6 months to 4 years.

Using the above assumption, a guideline for soil ingestion, dermal contact and inhalation, which applies to agricultural (residential use of farm property only), urban residential and park (playground) soil can be determined as:

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

where,

SQG_{DH} = direct human health-based soil quality guideline (mg/kg)
TDI = tolerable daily intake for toddler = 0.48 mg/kg bw/day (Section 5.8.1; 7.3)
EDI = estimated daily intake for toddler (median) = 0.55 mg/kg bw/day (Appendix VII)
SAF = soil allocation factor of 0.20 (CCME 2006)
BW = body weight for toddler = 16.5 kg (CCME 2006; Appendix IV)
SIR = soil ingestion rate for toddler = 0.00008 kg/day (CCME 2006; Appendix V)
IR_S = soil inhalation rate for toddler = 6.3 x 10^{-9} kg/day [*i.e.* inhalation rate for toddler = 8.3 m^3/d x suspended soil dust concentration of 7.6 x 10^{-10} kg/m^3 (Allan *et al.* 2008; HC 2010b)]
SR = soil dermal contact rate for toddler = 0.000069 kg/d [surface area of hands of 0.043 m^2 x soil adherence factor of 0.001 kg/m^2-event + surface of arms and legs of 0.258 m^2 x soil adherence factor of 0.0001 kg/m^2-event] (CCME 2006);
AF_G = relative absorption factor for zinc across the gut = 1.0 (by default)
AF_L = relative absorption factor for zinc across lung tissue = 1.0 (by default)
AF_S = relative absorption factor for zinc across the skin = 0.1 (Section 7.4)
ET_1 = exposure term 1 (unitless) – 7 days per week/7 x 52 weeks per year/52 at the site = 1.0 (CCME 2006)
ET_2 = exposure term 2 (unitless) – 24 hours per day/24 at the site = 1.0 (CCME 2006)
BSC = background soil concentration = 48.1 mg/kg (Grunsky 2010a)

The exposure term ET_1 in the above equation is the ratio of the defined exposure period for each land use to the maximum exposure period (24 hours/day x 365 days/year). Note that hours per day exposure (ET_2) is applied to soil inhalation, but is not considered for soil ingestion or dermal contact, consistent with HC (2008; 2010b) recommendations, since soil ingestion and dermal contact are not expected to occur at a uniform rate throughout the day.

As shown above, the background soil concentration is added back into the equation to calculate the SQG. It is initially removed when the exposure resulting from it is subtracted out along with the estimated daily intake. As the median estimated daily intake of zinc, as total zinc (i.e., EDI = 0.55 mg/kg bw/day) is greater than the TDI for total elemental zinc (i.e., TDI = 0.48 mg/kg bw/day), the human health soil quality guideline derived using the CCME (2006) protocol would be set to the background soil concentration of zinc (i.e., 48.1 mg/kg).

Recently, alternative approaches to address issues when the EDI approaches or exceeds the TDI have been considered (Appendix XVIII). Under such circumstances, it has been suggested that soil quality guidelines (and risk assessments) could be based on an acceptable level of exposure that would be equal to the lower of the following: 20 per cent of the TDI or 10 per cent of the EDI.

For zinc, "10 per cent of the EDI" (i.e., 0.055 mg/kg bw/day for toddlers) represents a more conservative value than "20 per cent of the TDI" (i.e., 0.096 mg/kg bw/day) for the toddler age group. (For adults, considered for industrial land use guidelines, the EDI does not exceed the TDI and, thus, this adjustment is not applicable). Consequently, under this approach, the value of "10 per cent of the EDI" is recommended for use in the derivation of the soil quality guideline for zinc for agricultural and residential land uses (see equation 2 in Appendix XVIII). Consequently, the SQG_{DH} may be modified to:

$$SQG_{DH} = \frac{(0.1 \times EDI) \times BW}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S) \times ET_2] \times ET_1}$$

where,

SQG_{DH} = direct human health-based soil quality guideline (mg/kg)
EDI = estimated daily intake for toddler (median) = 0.55 mg/kg bw/day (Appendix VII)
BW = body weight for toddler = 16.5 kg (CCME, 2006, Appendix IV)
SIR = soil ingestion rate for toddler = 0.00008 kg/day (CCME 2006, Appendix V)
IR_S = soil inhalation rate for toddler = 6.3 x 10^{-9} kg/day [*i.e.*, inhalation rate for toddler = 8.3 m^3/d x suspended soil dust concentration of 7.6 x 10^{-10} kg/m^3 (Allan *et al.* 2008; HC 2010b)
SR = soil dermal contact rate for toddler = 0.000069 kg/d [hands surface area of 0.043 m^2 × soil adherence factor of 0.001 kg/m^2-event + arms/legs surface area of 0.258 m^2 × soil adherence factor of 0.0001 kg/m^2-event (CCME 2006)]
AF_G = relative absorption factor for zinc across the gut (1.0, by default)
AF_L = relative absorption factor for zinc across lung tissue (1.0, by default)
AF_S = relative absorption factor for zinc across the skin (0.1, Section 7.4)
ET_1 = exposure term 1 (unitless) – 7 days per week/7 x 52 weeks per year/52 at the site (1.0) (CCME 2006)
ET_2 = exposure term 2 (unitless) – 24 hours per day/24 at the site (1.0) (CCME 2006)

Therefore, under the above approach, the SQG_{DH} for zinc in soil at agricultural and residential/parkland sites is calculated to be 10,350 mg/kg; rounded to two significant figures, the SQG_{DH} is 10,000 mg/kg.

The agricultural and residential/parkland SQG_{DH} for zinc based on the 10 per cent of the EDI approach is recommended for use. Although it provides a greater agricultural and residential/parkland SQG_{DH} for zinc than defaulting to the background soil concentration or practical limit of quantification, (as prescribed in CCME (2006) when the EDI>TDI), the agricultural and residential/parkland SQG_{DH} developed using the 10 per cent of the EDI approach is scientifically defensible and will be adequately protective of Canadians' health. This is quite clear when considering that the UL for zinc is much greater than 10 per cent of the EDI value. Consequently, the agricultural and residential/parkland SQG_{DH} for zinc in soil at agricultural and residential/parkland sites is recommended to be 10,000 mg/kg**.**

7.5.2 Commercial Land Use

Commercial land sites are generically defined as sites at which commercial activities predominate. No manufacturing activities and no residential occupancy are expected to take place at commercial sites. A commercial site is fully accessible to all age classes, but it is used with less intensity, duration and frequency than a residential site. An example of a commercial site would be a typical urban shopping mall, which could contain a daycare.

For threshold contaminants, it is assumed that a toddler is the most sensitive receptor but with access restricted to 10 hours per day, 5 days per week and 48 weeks per year (e.g., daycare). Using the above assumptions, a guideline that applies to commercial soil can be determined using the same equations as those to determine the agricultural and residential/parkland values. Similar to the agricultural and residential/parkland approach, the SQG_{DH} for commercial lands was calculated using both the traditional CCME (2006) equation and then the 10 per cent EDI approach.

Using the traditional CCME (2006) approach, a SQG_{DH} of 48.1 mg/kg for zinc in soil would be set, based on the background soil concentration. On the other hand, using the 10 per cent of the EDI approach, the SQG_{DH} for zinc in soil at agricultural and residential/parkland sites is calculated as 15,698 mg/kg; 16,000 mg/kg when rounded to two significant digits.

The SQG_{DH} for zinc based on the 10 per cent of the EDI approach is recommended for use. Although the EDI approach provides a greater SQG_{DH} for zinc than the CCME equation, the SQG_{DH} developed using the 10 per cent of the EDI approach is scientifically defensible and will be adequately protective of Canadians' health. Consequently, the recommended SQG_{DH} for zinc in soil at commercial sites is 16,000 mg/kg**.**

7.5.3 Industrial Land Use

Industrial lands typically have limited or restricted access to the public so that adult occupational exposure will predominate. The typical exposure period for an adult at an industrial site is assumed to be 10 hours per day, 5 days per week and 48 weeks per year. For industrial land use, only adult receptors are considered. The industrial soil guideline is derived using the equation from the CCME protocol (2006) and as shown in Section 7.5.1:

where,

TDI = tolerable daily intake = 0.57 mg/kg bw/day (Sections 5.8.1, 7.3)
EDI = estimated daily intake for an adult (median) = 0.18 mg/kg bw/day (Appendix VII)
BW = body weight for an adult = 70.7 kg (CCME 2006, Appendix IV)
SIR = soil ingestion rate for an adult = 0.00002 kg/day (CCME 2006, Appendix V)
SR = soil dermal contact rate for an adult = 1.14×10^{-4} kg/d [hands surface area of 0.089 m^2 x soil adherence factor of 0.001 kg/m^2-event plus arms surface area of 0.25 m^2 x soil adherence factor of 0.0001 kg/m^2-event (CCME 2006)]
IR_S = soil inhalation rate for an adult = 1.3 x 10^{-8} kg/d [*i.e.*, inhalation rate for an adult = 16.6 m^3/d x suspended soil dust concentration of 7.6 x 10^{-10} kg/m^3 (Allan *et al.* 2008; HC 2010b)]
ET_1 = exposure term 1 = 0.66 [5 days per week/7 x 48 weeks per year/52 at the site (CCME 2006)]
ET_2 = exposure term 2 = 0.42 [10 hours per day/24 at the site (CCME 2006)]

Therefore, the human health soil quality guideline (SQG_{DH}) for zinc on industrial lands was calculated as 265,394, and 270,000 mg/kg in dry soil for industrial sites (rounded). Following calculation of the off-site migration check value, the industrial SQG_{HH} is adjusted to 140,000 mg/kg (see Section 7.8).

7.6 Protection of Groundwater Used as a Source of Raw Water for Drinking

No guideline for protection of groundwater used as a source of raw water for drinking was derived for zinc (as total) due to constraints on the mathematical model when applied to inorganic substances (CCME 2006).

7.7 Guideline for Consumption of Produce, Meat and Milk

The check mechanism for consumption of produce, meat and milk was not required because zinc is an essential element that living organisms naturally concentrate and therefore accumulation within an organism is not indicative of toxicity (WHO 2001). Many organisms, including humans, can auto-regulate zinc concentrations and higher-than-normal exposures may or may not lead to higher-than-normal body burdens (ATSDR 2005) without adverse health effects.

7.8 Off-site Migration Guidelines for Commercial and for Industrial Land Uses

In the derivation of soil quality guidelines for commercial and industrial sites, only exposure scenarios for on-site exposure are considered. Transfers of contaminated soil from one property to another are possible by environmental occurrences such as wind and water erosion (CCME 2006).

The Universal Soil Loss Equation and the Wind Erosion Equation are utilized to estimate the transfer of soil from one property to another. The following equation allows the calculation of the concentration of eroded soil from the site that will raise the contaminant concentration in the receiving soil to equal the agricultural guideline within a specific time frame. This concentration is referred to as the human health soil quality guideline for off-site migration ($SQG_{OM\text{-}HH}$). If the human health soil quality guidelines, SQG_{HH}, for commercial or industrial sites are found to be above $SQG_{OM\text{-}HH}$, then the adjacent property could potentially become unacceptably contaminated from off-site deposition (CCME 2006) and may need to be adjusted according to the $SQG_{OM\text{-}HH}$. The following equation has been derived to allow the calculation of $SQG_{OM\text{-}HH}$.

$$SQG_{OM-HH} = 14.3 \times SQG_A - 13.3 \times BSC$$

where,

$SQG_{OM\text{-}HH}$ = human health soil quality guideline for off-site migration (i.e., the concentration of contaminant in eroded soil (mg/kg))

SQG_A = human health soil quality guideline (SQG_{HH}) for agricultural land use (10,000 mg/kg)

BSC = background concentration of zinc in the receiving soil (48.1 mg/kg)

The $SQG_{OM\text{-}HH}$ for commercial and for industrial land uses was determined to be 140,000 mg/kg (to two significant digits), which is greater than the SQG_{DH} for commercial (16,000 mg/kg), but less than the SQG_{DH} for industrial (270,000 mg/kg) land uses. Therefore, the industrial SQG_{HH} was modified to protect against off-site migration at this land use and is therefore set at 140,000 mg/kg.

7.9 Final Human Health Soil Quality Guidelines

Based on the CCME (2006) protocol, three types of exposure pathways are evaluated: required pathways, applicable pathways and check mechanisms. The exposure pathways evaluated and the SQGs for each of the pathways derived are listed in Table 5 below.

Human health SQGs were derived for zinc at contaminated sites, based on ingestion, inhalation and dermal contact with contaminated soil (SQG_{DH}). The proposed human health SQGs for agricultural and residential/parkland uses is 10,000 mg/kg. For commercial land uses, the proposed human health soil quality guideline is 16,000 mg/kg and, for industrial land uses, the proposed human health soil quality guideline is 140,000 mg/kg soil. The industrial SQG_{HH} is based on a check mechanism for off-site migration of eroded soil from industrial land use deposited onto adjacent more sensitive lands.

While the SQG_{DH} provided above are considered to be protective at most sites, not all exposure pathways have been evaluated, such as zinc levels in garden produce, meat, milk consumption (SQG_{FI}) or zinc leaching from soils to groundwater or surface water used for drinking water (SQG_{PW}). A site-specific risk assessment and sampling of additional media may be warranted at sites where these media may be affected by elevated levels of zinc in soil.

With the above in mind, the SQG_{DH} are considered to be protective of human health at most sites.

8.0 RECOMMENDED CANADIAN SOIL QUALITY GUIDELINES

According to the soil protocol (CCME 2006), both environmental and human health soil quality guidelines are developed for four land uses: agricultural, residential/parkland, commercial and industrial. The environmental soil quality guidelines for zinc, presented in Chapter 6, were considered along with the human health guidelines presented in Chapter 7 in making final recommendations for Canadian Soil Quality Guidelines for the protection of environmental and human health (CCME 2006). The lowest value generated by the two approaches (environmental and human health) for each of the four land uses is recommended by CCME as the Canadian Soil Quality Guidelines. These values, along with previous recommended Canadian Soil Quality Guidelines for the protection of environmental and human health (CCME 1999) are presented below in Table 5. The interim remediation criteria (CCME 1991) are also presented for comparison purposes.

Table 5: Canadian Soil Quality Guidelines for Zinc (mg/kg)

	Land use			
	Agricultural	**Residential/ parkland**	**Commercial**	**Industrial**
Soil Quality Guideline	**250**[i]	**250**[i]	**410**[i]	**410**[i]
Human health guidelines/check values				
SQG_{HH}	10,000	10,000	16,000	140,000
Direct contact guideline[ii]	10,000	10,000	16,000	270,000
Inhalation of indoor air check[iii]	NC	NC	NC	NC
Off-site migration check	–	–	140,000	140,000
Groundwater check (drinking water)[iv]	NC	NC	NC	NC
Produce, meat, and milk check[v]	NC	NC	–	–
Environmental health guidelines/check values				
SQG_{E}	250	250	410	410
Soil contact guideline	250	250	450	450
Soil and food ingestion guideline	980	–	–	–
Nutrient and energy cycling check	280	280	410	410
Off-site migration check	–	–	2,900	2,900
Groundwater check (aquatic life)[iv]	NC	NC	NC	NC
Zinc Soil Quality Guideline (CCME 1999)	200	200	360	360
Interim soil quality criterion (CCME 1991)	600	500	1,500	1,500

Notes:

NC = not calculated; the dash indicates a guideline/check value that is not part of the exposure scenario for this land use and therefore is not calculated.

[i]Data are sufficient and adequate to calculate an SQG_{E} and an SQG_{HH}. Therefore the soil quality guideline is the lower of the two, and supersedes the 1999 soil quality guidelines and the 1991 interim soil quality criterion.

[ii]The direct human health-based soil quality guideline is based on direct exposure to soil ingestion, dermal contact and particulate inhalation.

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

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

[v]This check is intended to protect against chemicals that may bioconcentrate in human food. Zinc is not expected to exhibit this behaviour, therefore this pathway was not evaluated.

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Appendix I. Available Data on Zinc Concentrations in the Environment

Air

Location	Year	Mean Concentration µg/m³ (SD)	Range µg/m³	Comments	Reference
Canada	2003–2009	0.013 (0.014)	0–12	Based on NAPS data (2003–2009)	HC 2011a
Canada	–	<1	–	estimated background	EC 1999a
Eight Canadian Cities	1986–96	0.0258	–	only $PM_{2.5}$	Burnett *et al.* 2000
Ontario – Windsor	2004	1,189 µg/g	262–2,651 µg/g	$PM_{2.5}$	Rasmussen *et al.* 2007
	2005	0.240	0.085–0.382	$PM_{2.5}$ measured over 24hrs by ICP–MS	Niu *et al.* 2009
	2005	0.121	0.049–0.217	$PM_{2.5}$ measured over 2wks by ICP–MS	Niu *et al.* 2009
Ontario – Burnt Island	1992–94	0.0066497	–	rural	Biegalski and Hopke 2004
Ontario					
• Southern		0.007			
• Central		0.013			
• Northern	1982	0.007			Chan *et al.* 1986
Vermont, USA	1998	34.31 µg/L	3.4–156.38	cloud water samples	Malcolm *et al.* 2003
General		Under 0.5			Eisler 1993

Note: Data from shaded row was used to calculate EDIs for Zn.

Indoor Air

Location	Year	Mean Concentration µg/m³	Range µg/m3	Comments	Reference
Alberta – High Level	1997	0.0139		2 houses sampled per day over 10 days ($PM_{2.5}$)	Alberta Health 1998
Ontario – Windsor	2004	440 µg/g	165–1,196 µg/g	$PM_{2.5}$	Rasmussen *et al.* 2007

Location	Year	Mean Concentration µg/m^3	Range µg/m3	Comments	Reference
United States – Baltimore, Maryland	1998	0.02686		Median indoor central aerosol ($PM_{2.5}$)	Graney *et al.* 2004
United Sates – Minneapolis/St. Paul, MN	1999	0.0108		$PM_{2.5}$	Adgate *et al.* 2007
United States – Chicago, IL	1994–1995	0.0295	nd – 0.088	$PM_{2.5}$	Van Winkle and Scheff 2001
Belgium – Antwerp	n/a	0.0173	0.002–0.056	$PM_{2.5}$	Stranger *et al.* 2009
Singapore	2004	0.0417		Living room ($PM_{2.5}$)	Balasubramanian and Lee 2007
		0.0437		Master bedroom ($PM_{2.5}$)	
		0.0318		Bedroom ($PM_{2.5}$)	
Sweden – Göteborg	2002–2003	0.017	0.0064–0.04	$PM_{2.5}$	Molnar *et al.* 2006
Taiwan – Taipei	1992–1993	0.228		Average of 3 residences	Li *et al.* 1993
		0.182		Residence 1	
		0.386		Res 2 (near iron foundry)	
		0.115		Res 3 (near iron foundry)	

Note: Data from shaded rows was used to calculate EDIs for Zn

Soil

Location	Year	Soil Type	Sample Depth	Mean Concentration (SD) mg/kg	Range mg/kg	Comments	Reference
Canada		Glacial tills		48.1 (48.4)	<2–1,770		Rencz *et al.* 2006, Grunsky 2010a
Canada, USA and Mexico	2008–2009		PH	60		Median values, <2mm fraction (not milled), analyzed by ICP-MS/AES following partial/aqua regia	Friske *et al.* 2014
			0–30cm	48			
			A	60			
			B	53			

Location	Year	Soil Type	Sample Depth	Mean Concentration (SD) mg/kg	Range mg/kg	Comments	Reference
			C	49		digestion	
Canada	–	uncultivated soils	A, B and C horizons	74 (NR)	10–200		
• Appalachian Region				81 (NR)			
• Canadian Shield				54 (NR)			
• St. Lawrence Lowlands				80 (NR)			
• Interior Plains				64 (NR)			
• Cordilleran Region				73 (NR)		uncontaminated, remote from ore bodies	McKeague and Wolynetz 1980
Canada – overall	–			76	6.3–360	background samples	Sheppard *et al.* 2007
Canada – overall	–				91.5–431.2		Impelliteri *et al.* 2003
Canada – New Brunswick, Nova Scotia, Prince Edward Island			Public Health (PH) layer	51.5(37.5)	6.3–245.2		
			A horizon	56.3(39.1)	3.8–249.3		
			B horizon	64.5(33.6)	5–226.4		
			C horizon	67.6(34.5)	19.8–286.3	non-urban soils	Grunsky 2010b
Canada – Prairies				74.13	20–258		Geological Survey of Canada 2004
Canada – southeastern					0.012–1.343		
		forests			0.31–5.093	bulk samples	Courchesne *et al.* 2006
		rhizosphere samples				rhizosphere samples	
BC – Mt Robson Provincial Park	2002				19.4–77.5	disturbed samples	
					15.5–137	undisturbed samples	Arocena *et al.* 2006

Location	Year	Soil Type	Sample Depth	Mean Concentration (SD) mg/kg	Range mg/kg	Comments	Reference
BC – Trail	–	undisturbed surface soil	0–15 cm	94.1(45)	32–186	background samples	Sanei *et al.* 2007
Saskatchewan	2003	agricultural area			73.71–106.78	control site for the study	Lipoth and Schoenau 2007
Manitoba – Flin Flon	–	forested soils	LFH	98 (6)	–	control soil, 68.2 km from a copper–zinc smelter and mine	Hogan and Wotton 1984
			0 to 5 cm	90(6)			
			5 to 10 cm	100 (6)			
			10 to 15 cm	80 (6)			
Manitoba – Winnipeg	–	urban soils	surface	96 (NR)	62–116	urban soils	Mills and Zwarich 1975
Ontario – forest floors	–		surface	2.5	–		Watmough *et al.* 2004
Ontario – Halton	–	agricultural soils	surface	126 (89)	50–821	sludge treated	Webber and Shamess 1987
				113 (34)	57–243	background samples	
Ontario – Port Colborne	2002				68.4–186	contaminated soils	Everhart *et al.* 2006
Ontario – south central	–	mineral			12.5–128		Watmough *et al.* 2004
Ontario – southwestern	–	agricultural soils	Ap	88 (28)	40–163		
			B	87 (29)	35–140		
			C	71 (26)	40–128	uncontaminated soils	Whitby *et al.* 1978
Ontario – Sudbury	1995			17–95	–		Gratton *et al.* 2000
Ontario – Sudbury	–	organic	FH horizon	184	–	uncontaminated soils	Johnson and Hale 2004
Ontario – Sudbury	–			–	11.2–64.13		Nkongolo *et al.* 2008
Ontario – Sudbury (and Rouyn–Noranda)	2001			–	34–120		Feisthauer *et al.* 2006
Ontario		Sandy Soil		40			Adriano 2001

Location	Year	Soil Type	Sample Depth	Mean Concentration (SD) mg/kg	Range mg/kg	Comments	Reference
		Loam		64			
		Clayey		62			
		Organic Soil		66			
Ontario	1994			120		Rural parkland – includes areas near Sudbury known for metal mining	OMEE 1993
	1994			140		Urban parkland – includes areas near Sudbury known for metal mining	OMEE 1993
Ontario				76.46	22–320		Geological Survey of Canada 2004
Québec – Montreal	–			183	–	gardens	Marr *et al.* 1999
Québec – Montreal Island	1997			489	203–1350	urban soils	Ge *et al.* 2000
Québec – Mt Richardson	1996	sub–alpine	FH horizon	44	–		
		sub–alpine	surface litter	73	–		Evans *et al.* 2005
Québec – Rouyn–Noranda	–	organic	FH horizon	100	–	uncontaminated soils	Johnson and Hale 2004
Worldwide	–	various soils			10–300		He *et al.* 2005
Worldwide	–	outdoor dust	surface/air		800–1,600		Kim and Fergusson 1993
Worldwide		Igneous rock		65	5–1,070		
		Limestone		20	<1–180		
		Sandstone		30	5–170		
		Soil		90	1–900		Adriano 2001
Earth Average				70			CMBEEP 1979

Note: Data from shaded rows was used to represent Canadian background and to calculate EDIs for Zn.

Dust

Location	Year	Sample location	Mean Concentration (SD) mg/kg	Range mg/kg	Comments	Reference
Ontario – Ottawa	2001–02	Indoor dust	793	182–2,491		Rasmussen *et al.* 2008
		Outdoor dust	119	27 –484		Rasmussen *et al.* 2008
Ontario – Ottawa	1993	Indoor dust	716.9	239.1–1,840	50 homes from 10 neighbourhoods	Rasmussen *et al.* 2001
Ontario – Toronto		Outdoor dust	6,000			VanLoon 1973
United States (NHEXAS) Arizona	1995–97	Indoor dust	1,175.8 (1,595.6)	238.2–13,640		US EPA 2009
United States – New Jersey	1992–94	Indoor dust PM_{10}	2,223 (173)	–	Concentrations converted from mass %.	Adgate *et al.* 1998
Australia – Sydney	1999	Indoor dust	657 (1,140)	100–9,930	82 homes from 6 suburbs	Chattopadhyay *et al.* 2003
Australia – Sydney	1997 & 99	Indoor dust	23,673 (15,931)	1,630	<500m from industrial site (n=10)	Davis and Gulson 2005
			2,825 (6,662)	598	500–1,500m from industrial site (n=19)	
			603 (215)	396	>1500m from industrial site (n=8)	
New Zealand – Christchurch	1987	Indoor dust	21,700 (35,400)	871–205,000		Kim and Fergusson 1993
United Kingdom	2005	Indoor dust	666 (240)	213–1,300		Turner and Simmonds 2006
Germany	1990–92	Indoor dust	635	NR–30,600		Seifert *et al.* 2000
Poland – Warsaw	1997	Indoor dust	1,430 (813)	534–4,080	63–125µm	Lisiewicz *et al.* 2000
			1,230 (487)	592–2,450	32–63µm	
			1,200 (376)	609–2,100	0–32µm	
International – Persian Gulf			64.4			Madany *et al.* 1994

Location	Year	Sample location	Mean Concentration (SD) mg/kg	Range mg/kg	Comments	Reference
Bahrain	–	Indoor dust	202 (97.8)	31–475		Madany *et al.* 1994
Oman – Muscat	2004	Indoor dust	753 (1,162)	18–8,504		Yaghi and Abdul-Wahab 2004
Worldwide	–	Outdoor dust	800–1,600	–		Kim and Fergussen 1991

Note: Data from shaded rows was used to calculate EDIs for Zn.

NR=not reported

Sediment

Location	Year	Mean Concentration (SD) mg/kg dw	Range mg/kg dw	Comments	Reference
Canada	–	90	–	baseline	NRC 1979
Yukon Territory	–	139.58	–		Gamberg *et al.* 2005
BC – Fraser River Basin (5 lakes)					
Moose Lake		86.6 (6.4)			
Stuart Lake		117(8.5)			
Chilko Lake		164(16)			
Kamloops Lake		147(13)			
Nicola Lake		87(6.6)		background (pre-1900)	
Harrison Lake		181.9(9.9)		calculated from bottom 3–4	
All lakes	1993–94		77–213	samples from each core ± 1 SD	Gallagher *et al.* 2004
BC – Howe Sound	–	–	200–357	mining area	EC 1999a
BC – Upper Columbia River			45–51		Johnson *et al.* 1990 in ATSDR 2005
Great Lakes					
Lake Superior		103	9.6–176		
Lake Huron		80.7	8.2–248		
Georgian Bay	2001–02	114	31.1–20.7	n=87	Gewurtz *et al.* 2008
Lake Erie	1997–98	88.3	–	background	Marvin *et al.* 2004
Lake Ontario	–	102.9	–	background	
Lake Erie/Ontario	2001	–	116–603		Marvin *et al.* 2007
Ontario – 12 lakes	1998	–	29.5–120.6	near shore sediment	Shuhaimi-Othman *et al.* 2006
Ontario – Killarney Park	–	<300	–	remote area	Belzile *et al.* 2004
Ontario – stormwater management facilities	–	–	95–11,16		Marsalek *et al.* 2006
Ontario – Sudbury	2001	13.1–393.9	–	lake samples	Pyle *et al.* 2005
Québec and Ontario		152.2	3–559.9	187 QC lakes and 52 ON lakes	Rowan and Kalff 1993 in ATSDR 2005
Québec – Montreal (St. Lawrence fluvial lakes)	2004–05	–	31–550		Desrosiers *et al.* 2008
New Brunswick – Nepisiguit River	–	–	447	mining area	EC 1999a
Bay of Fundy	1997–2002	66	33–100		Hung and Chmura 2007
Bay of Fundy	1999	35.1–69.4	–		Chou *et al.* 2000

Drinking Water

Location	Year	Mean Concentration (SD) µg/L	Range µg/L	Comments	Reference
Canada – Ontario, Newfoundland and Saskatchewan	1998–2009	11.1 (49.1)	–	n = 14,714	HC 2011b
Ontario	1998–2007	5.90 (21.5)	–	n = 3,812	HC 2011b
Saskatchewan	2000–2009	11.4 (48.6)	–	n = 8,379	HC 2011b
Newfoundland and Labrador	2000–2009	18.0 (73.4)	–	n = 2,523	HC 2011b
Canada	1995–1996	2.2	1.6–4.9	tap water	Dabeka *et al.* 2002
Canada – Nova Scotia, Ontario, Saskatchewan	–	26.5	1–54	treated drinking water	HC 1987
Canada	1980–1985				
Pacific		–	1–30		
Western		–	1–29		
Central		–	1–70		
Atlantic		–	1–19		
Drinking water		10	–	treated water	HC 1987

Note: Data from shaded rows was used to calculate the EDI for Zn.

Surface Water

Location	Year	Mean Concentration µg/L	Range µg/L	Comments	Reference
Canada – soil pore water	–	140	–	background (95th percentile)	Doyle *et al.* 2003
Canada – surface water	–	12	–	background (95th percentile)	
BC	1998	3.2	–	Cowichan River	Rideout *et al.* 2000
		0.003		Koksilah River	
	2005–2009	<1	7.7 (max)	Cowichan and Koksilah Rivers	Dessouki 2010
Ontario – 12 lakes	1998	–	2.2–10.4	surface water, impacted lakes 6–	Shuhaimi-Othman *et al.*

				12 km from smelter	2006
	1998		2.74–4.4	intermediate lakes 32–52 km from smelter	Shuhaimi-Othman *et al.* 2006
	1998		1.6–2.8	control lakes 94–154 km from smelter	Shuhaimi-Othman *et al.* 2006
Ontario – Sudbury	1983–1995	6.5	1.9–17.9	lake samples	Mallory *et al.* 1998
Ontario – Sudbury	2001	1.0–17.7	–	lake samples	Pyle *et al.* 2005
Lake Ontario – tributaries	1997–1998	–	NS – 8.71		OMOE 1999
Lake Ontario	May–June 1993	1.6	1.30–3.26	n=59; mean for May–June & Oct.	Nriagu *et al.* 1996
Lake Ontario	Oct. 1993	1.6	0.56–2.72	n=47 mean for May–June & Oct.	Nriagu *et al.* 1996
Lake Erie	Aug 1993	0.87	0.22–3.77	n=32	Nriagu *et al.* 1996
Lakes Erie and Ontario			0.003–0.11		Coale and Flegal 1989 in ATSDR 2005
Lake Superior	Sept. 1991	2.77	1.44–8.67	n=47	Nriagu *et al.* 1996
United States	–	–	20–50	ambient water	ATSDR 2005
Vermont – USA	1998	4.01	0.31–16.33	rain samples	Malcolm *et al.* 2003

Groundwater

Location	Year	n	Mean Concentration µg/L 97.5%ile	Range µg/L	Comments	Reference
Ontario	2002–2007	419	159	0.5–274	Provincial Groundwater Monitoring Information System (PGMIS)	OMOE 2011a
Ontario	1997, 1999–2002	747	42.06	0.2–159	Drinking Water Surveillance Program (DWSP)	OMOE 2011a

Commercial Foods

Food Type	Year	Mean Concentration	Range	Comment	Reference

Food Type	Year	Mean Concentration	Range	Comment	Reference
Canada	1995–1996	3.49 µg/L	1.61–18 µg/L	retail distilled water	Dabeka *et al.* 2002
		8.1 µg/L	1.61–117 µg/L	retail mineral water	
		12 µg/L	0.9–234 µg/L	retail spring water	
Canada: Total Diet Studies					
Seeds	2003–2007		48,266–54,692 ng/g	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2011b
Meat, poultry or eggs	2003–2007		12,081–28,065 ng/g	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2011b
Herbs and spices	2003–2007		10,773–19,768 ng/g	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2011b
Whole wheat bread	2003–2007		9,943–16,489 ng/g	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2011b
infant formula; milk basedn	2003–2007		6,460–10,880 ng/g	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2011b
infant formula; soy-based	2003–2007		4,294–8,277 ng/g	Montreal, Winnipeg, Toronto, Halifax, Vancouver data	HC 2011b

Human milk for breast fed infants

Location	n	Mean Concentration (SD) µg/L	Range µg/L	Comment	Reference
Canada - Newfoundland	17	Median: 4,850 (520)	3–28	Colostrum-full-term gestation	
	119	Median: 2,595 (1445)		Mature milk-full-term gestation	
	24	Median: 5,310 (2490)		Colostrum-preterm gestation	
	128	Median: 2,160 (1558)		Mature milk-preterm gestation	Friel *et al.* 1999

Location	n	Mean Concentration (SD) µg/L	Range µg/L	Comment	Reference
United States – Colorado	259		4,000–44,500	1–28 days postpartum. No overall mean reported. See HC 2011a for data per sampling date.	Casey *et al.* 1985
Austria	55	3840	110–13,160	1–293 days postpartum	Krachler *et al.* 1998
	27	10,120 (5,080)	2,960–23,090	2–3 day postpartum colostrum	Krachler *et al.* 1999
Italy – Turin	30	3,080 (508)	–	Mature milk 2 mths postpartum	Coni *et al.* 2000
Italy	36	3,420 (2,408)	710–11,730	Mix of urban and rural smokers and non-smokers	Coni *et al.* 1990
Portugal	34	12,137 (4,714)	1,869–22,050	Colostrum	Almeida *et al.* 2008
	19	2,785 (1,205)	391–50,88	Mature milk	
	31	4,011 (1,178)	2,073–6,781	Mature milk (7 days)	
	31	2,161 (589)	1,218–6,781	Mature milk (4 weeks)	
	31	1,491 (620)	32–2,956	Mature milk (8 weeks)	
	31	1,084 (537)	206–2,195	Mature milk (12 weeks)	
	31	1,014 (462)	155–2,216	Mature milk (16 weeks)	Matos *et al.* 2009
Japan	68	5,320 (1,780)	2,730–11,600	Mature milk 5–8 days post partum	Honda *et al.* 2003
Japan – Tokyo	27	5,530 (2,180)		Mature milk 6–9 days postpartum	Li *et al.* 1990
	27	5,740 (1,950)		Mature milk 6–9 days postpartum	Li *et al.* 1989
United Arab Emirates	120	2,730 (1,150)	390–6,150		Kosanovic *et al.* 2008
	205	1,468 (1,093)	115–6,282	Mature milk 4–80 weeks	Abdulrazzaq *et al.* 2008
Brazil – Rio de Janiero	48	6,970 (2,820)	2,410–14,470	Colostrum 1–4 days postpartum	daCosta *et al.* 2002

Note: Data in shaded rows was used to calculate the EDIs for Zn.

Consumer Products

Location	Product	Year	Concentration mg/kg	Range mg/kg	Comment	Reference
Canada	Dietary supplements	2004–05		5.1–50	Supplements of marine origin available in Canada	Leblond *et al.* 2008

Unknown	Paint	–	31.10			Mielke *et al.* 2001
Unknown	Coal	–	13	–		Finkelman 1999

Human Tissues and Biological Fluids

Location	Tissue	Year	Concentration	Range	n	Comment	Reference
Canada	Blood	2000s	6.37 µg/L		5319	Aged 6–79 yrs	HC 2010a
	Urine	2000s	307 µg/g creatinine		5479	Aged 6–79 yrs	HC 2010a
Canada – BC	Urine	2000s	285.43 µg/g creatine			Aged 30–65 yrs	Clark *et al.* 2007
United States	Fingernails, toenails and hair			94–129 µg/g			ATSDR 2005
	Cadaver tissues			1.5–55 µg/g			ATSDR 2005

Biota

Species and Location	Year	Concentration (µg/g dry weight)	Range	Comments	Reference
Invertebrates					
Hyalella azteca (12 lakes in Ontario)	1998	–	77.5–272.6		Shuhaimi-Othman *et al.* 2006
Invertebrates (Chilliwack River, BC)	2000–01	228.6	–		Morrissey *et al.* 2005
Lobster Digestive Glands (Bay of Fundy)	1999	28–129 ww	–		Chou *et al.* 2000
Mussels (Atlantic Canada)	1991–99	55–123	–		Chou *et al.* 2003
Zebra Mussels (St. Lawrence River, Canada)	1996		140.3–340.3		deLafontaine *et al.* 2000
Snails (Elusive Lake, AK, USA)	1991–93	28.85	–		Allen-Gil *et al.* 1997
Fish					
Burbot muscle (Great Slave Lake)	1999–02	4	–		Evans *et al.* 2005
Burbot liver (Great Slave Lake)	1999–02	24.2	–		Evans *et al.* 2005
Unknown, liver (Great Slave Lake)	1999–02	34.5	–		Evans *et al.* 2005
Unknown, muscle (Great Slave Lake)	1999–02	3.1	–		Evans *et al.* 2005
Landlocked lake char (Arctic Canada)	1999–03	4.673–12.58 ww	3.5–17.0	Range of means from 5 lakes. Range is min and max from all data.	Muir *et al.* 2005
Pike liver (Great Slave Lake)	1999–02	46.9	–		Evans *et al.* 2005
Pike muscle (Great Slave Lake)	1999–02	4.2	–		Evans *et al.* 2005
Various fish (Chilliwack River, BC)	2000–01	87.76	–		Morrissey *et al.* 2005
Yellow perch liver (lakes near Sudbury, ON)	2001	99.1–1,441.0	–	Range of 12 means from 12 Sudbury, ON Lakes	Pyle *et al.* 2005
Yellow perch muscle (lakes near Sudbury, ON)	2001	23.5–276.9	–	Range of 12 means from 12 Sudbury, ON Lakes	Pyle *et al.* 2005
Trout liver (Elusive Lake, AK, USA)	1991–03	144.54	–		Allen-Gil *et al.* 1997

Species and Location	Year	Concentration (µg/g dry weight)	Range	Comments	Reference
Trout liver (Feniak Lake, AK, USA)	1991–03	169.82	–		Allen-Gil *et al.* 1997
Trout liver (Schrader Lake, AK, USA)	1991–03	89.13	–		Allen-Gil *et al.* 1997
Trout muscle (Elusive Lake, AK, USA)	1991–03	16.98	–		Allen-Gil *et al.* 1997
Trout muscle (Feniak Lake, AK, USA)	1991–03	16.22	–		Allen-Gil *et al.* 1997
Trout muscle (Schrader Lake, AK, USA)	1991–03	17.78	–		Allen-Gil *et al.* 1997
Other Aquatic Species					
Beluga whale liver (Canadian Arctic)	1998–2001	36.8 ww	18.5–53.2		Dehn *et al.* 2006
Beluga whale muscle (Canadian Arctic)	1998–2001	28.4 ww	16.3–66.7		Dehn *et al.* 2006
Bowhead whale liver (Canadian Arctic)	1998–2001	31.6 ww	6.99–135.11		Dehn *et al.* 2006
Bowhead whale muscle (Canadian Arctic)	1998–2001	33.85 ww	9.47–74.1		Dehn *et al.* 2006
Gray whale liver (Canadian Arctic)	1998–2001	29.7 ww	9.57–300.4		Dehn *et al.* 2006
Gray whale muscle (Canadian Arctic)	1998–2001	33.5 ww	19.1–74.8		Dehn *et al.* 2006
Harbor seal kidney (NL)	–	18.15–26.83 ww	–	Range of averages from 5 sites.	Veinott and Sjare 2006
Harbor seal liver (NL)	–	28.47–49.25 ww	–	Range of averages from 5 sites.	Veinott and Sjare 2006
Harbor seal muscle (NL)	–	16.89–24.13 ww	–	Range of averages from 5 sites.	Veinott and Sjare 2006
Birds					
Median ptarmigan kidney (across Canada)	1985–94	156.2	–		Pedersen and Lierhagen 2006
Median ptarmigan liver (across Canada)	1985–94	105.9	–		Pedersen and Lierhagen 2006
Eider species, kidney and liver (Canadian Arctic)	1997	–	120.5–188.2		Wayland *et al.* 2001
Seabird kidney (Canadian Arctic)	1983,91,93	106–199	84–254	Range of means from 5 species and 5 locations. Range indicates min and max from all data.	Braune and Scheuhammer 2008
Seabird muscle (Canadian Arctic)	1998–99	11.2–26.7 ww	–	Range of means from 8 species and 2 regions.	Borga *et al.* 2006

Species and Location	Year	Concentration (µg/g dry weight)	Range	Comments	Reference
Seabird liver (Canadian Pacific Coast)	1990	93.9–122.0	–	Range of means from 3 species at 4 sites.	Elliott and Scheuhammer 1998
Various species (near Chilliwack River, BC)	2000–01	131.8	–		Morrissey *et al.* 2005
Mammals					
Arctic hare muscle (SW Nunavut)	2003	67.2	–		Pedersen and Lierhagen 2006
Arctic hare muscle (SW Nunavut)	2003	96.71	–		Pedersen and Lierhagen 2006
Arctic hare muscle (SW Nunavut)	2003	192.73	–		Pedersen and Lierhagen 2006
Moose liver (YK)	–	34.87	–		Gamberg *et al.* 2005
Moose muscle (YK)	–	51.74	–		Gamberg *et al.* 2005
Adult elk kidney (Sudbury, ON)	1995–97	164.47	–		Parker and Hamr 2001
Adult elk liver (Sudbury, ON)	1995–97	74.77	–		Parker and Hamr 2001
Adult elk muscle (Sudbury, ON)	1995–97	70.65	–		Parker and Hamr 2001
Muskrat kidney (Sudbury, ON–contaminated)	–	43.47	–		Parker 2004
Muskrat liver (Sudbury, ON–contaminated)	–	64.17	–		Parker 2004
Muskrat kidney (North Bay, ON–uncontaminated)	–	44.82	–		Parker 2004
Muskrat liver (North Bay, ON–uncontaminated)	–	65.31	–		Parker 2004
Caribou and reindeer liver (Greenland)	1995–97	23.2–31.7 ww	–	Range of means from 4 sites.	Aastrup *et al.* 2000
Caribou and reindeer muscle (Greenland)	1995–97	17.5–39.6 ww	–	Range of means from 4 sites.	Aastrup *et al.* 2000
Plants					
Moss (Lower Fraser Valley BC, rural)	1993	19.3	–		Pott and Turpin 1998
Moss (Lower Fraser Valley BC, urban)	1993	43.2	–		Pott and Turpin 1998
Black spruce needles (Sudbury, ON)	–	–	3.48–21.08		Nkongolo *et al.* 2008
Pine (Sudbury, ON)	1995	10.4–66.4	–		Gratton *et al.* 2000

Species and Location	Year	Concentration (µg/g dry weight)	Range	Comments	Reference
Various plant forage species (Sudbury, ON)	1995–97		2.02–160.39	Data from 7 species at two sites.	Parker and Hamr 2001
Dandelions (Montreal, QC garden areas)	–	35.5	–		Marr *et al.* 1999
Dandelions (Montreal, QC industrial areas)	–	95	–		Marr *et al.* 1999
Dandelions (Montreal, QC parks)	–	71.4	–		Marr *et al.* 1999

Note:
NS = not stated
ww = wet weight

Appendix II. Yearly Average Intake of Zinc via Food Ingestion (Weight-adjusted) (Deterministic) mg/kg/day

Zinc (Zn)	0–6mo M&F	7mo–4yrs M&F	5–11yrs M&F	12–19yrs M&F	20+yrs M&F
2000	723.17	589.57	430.43	286.71	207.63
2001	772.76	553.34	390.94	256.08	178.99
2002	757.69	550.77	397.21	259.59	183.19
2003	711.37	529.77	365.02	229.86	162.52
2004	654.38	522.87	374.02	242.89	172.82
2005	709.94	533.82	379.73	249.32	177.10
2006	618.96	513.60	371.55	242.66	172.71
2007	792.36	573.72	402.23	261.55	184.13
Mean	717.58	545.93	388.89	253.58	179.89
Standard Deviation	58.67	26.02	21.22	16.98	13.15

Note:

– Reference: Dabeka *et al.* 2010. Above values were applied to non-breastfed infants for the purposes of calculating EDIs for infants (birth to 6 months). Breast milk concentrations were used to calculate the EDI for breastfed infants.

Appendix III. Typical Environmental Concentrations Used in EDI Calculations

Media	Units	Distribution	Statistics	Zinc
Drinking Water[1]	µg/L	Lognormal	Arithmetic Mean	11.09
			Standard Deviation	49.05
			Minimum	0
			Maximum	451
Outdoor Air[2]	µg/m^3	Lognormal	Arithmetic Mean	0.0133
			Standard Deviation	0.0138
			Minimum	0
			Maximum	0.12
Indoor Air[3]	µg/m^3	Lognormal	Arithmetic Mean	0.0148
			Standard Deviation	0.0168
			Minimum	0
			Maximum	0.15
Soil[4]	mg/kg	Lognormal	Arithmetic Mean	48.1
			Standard Deviation	48.4
			Minimum	0
			Maximum	417
Settled Dust[5]	mg/kg	Lognormal	Arithmetic Mean	1,278
			Standard Deviation	6,649
			Minimum	0
			Maximum	57,758
Breast Milk[6]	µg/L	Lognormal	Arithmetic Mean	3,581
			Standard Deviation	3,328
			Minimum	0
			Maximum	27,978

[1] Based on average zinc concentrations in drinking water from Ontario (1998–2007), Saskatchewan (2000–2009) and Newfoundland and Labrador (2000–2009) (HC 2011b).

[2] Outdoor air $PM_{2.5}$ concentrations NAPS data collected from 2003 to 2009 from British Columbia, Ontario, Québec and New Brunswick from urban and rural centres (HC 2011a).

[3] Indoor air concentrations based on $PM_{2.5}$ from: Alberta Health (1998); Adgate *et al.* (2007); Graney *et al.* (2004); Van Winkle and Scheff (2001); Stranger *et al.* (2009); Balasubramanian and Lee (2007); Molnar *et al.* (2006); (HC 2011a).

[4] Based on Geological Survey of Canada data from Grunsky (2010a), HC (2011a).

[5] Based on arithmetic mean of total barium in indoor settled dust from Rasmussen *et al.* (2001; 2008); NHEXAS 2010; Adgate *et al.* (1998); Chattopadhyay *et al.* (2003); Davis and Gulson (2005); Turner and Simmonds (2006); Seiffert *et al.* (2000); Kim and Fergusson (1993); Yaghi and Abdul-Wahab (2004); Lisiewicz *et al.* (2000); HC (2011a).

[6] Based on zinc concentration in breastmilk from Friel *et al.* (1999); Krachler *et al.* (1998; 1999); Casey *et al.* (1985); Coni *et al.* (1990; 2000); daCosta *et al.* (2002); Honda *et al.* (2003); Li *et al.* (1989; 1990); Almeida *et al.* (2008); Matos *et al.* (2009) and Abdulrazzaq *et al.* (2008); HC (2011a).

Appendix IV. Receptor Characteristics of the Canadian General Population[1]

	Statistic	Breastfed Infant (0 to 6 mo.)	Non-Breastfed Infant (0 to 6 mo.)	Toddler (7 mo. to 4 yr)	Child (5 to 11 yr)	Teen (12 to 19 yr)	Adult (20+ yr)
Body Weight (kg)	Minimum	2.8	2.8	7.1	14.2	30.0	38.1
	Maximum	21.5	21.5	35.9	71.5	112.2	126.5
	Mean	8.2	8.2	16.5	32.9	59.7	70.7
	Std. dev.	2.9	2.9	4.5	8.9	13.5	14.5
	Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Skin Surface Area Hands (cm^2)	Minimum	242	242	299	396	556	614
	Maximum	416	416	614	863	1,142	1,262
	Mean	320	320	430	590	800	890
	Std. dev.	30	30	50	80	100	110
	Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Skin Surface Area Arms (cm^2)	Minimum	200	200	396	797	1,409	1,588
	Maximum	1,367	1,367	1,882	2,645	3,465	3,906
	Mean	550	550	890	1,480	2,230	2,510
	Std. dev.	180	180	240	300	340	360
	Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Skin Surface Area Legs (cm^2)	Minimum	539	539	907	1604	3042	3753
	Maximum	1,496	1,496	3,012	5,655	7,945	8,694
	Mean	910	910	1,690	3,070	4,970	5,720
	Std. dev.	160	160	340	660	810	760
	Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Soil Loading to Exposed Skin[2] Hands Surfaces other than Hands (kg/cm^2/event)	Default	1.0×10^{-7} 1.0×10^{-8}	1.0×10^{-7} 1.0×10^{-8}	1.0×10^{-7} 1.0×10^{-8}	1.0×10^{-7} 1.0×10^{-8}	1.0×10^{-7} 1.0×10^{-8}	1.0×10^{-7} 1.0×10^{-8}
Time Spent[3] Outdoors (hr/d)	Minimum	0.000	0.000	0.000	0.000	0.000	0.000
	Maximum	3	3	3	4	9.45	10.76
	Mean/Mode	1.25	1.25	1.25	2.2	1.42	1.43
	Std. dev.	N/A	N/A	N/A	N/A	1.17	1.28
	Distribution	Triangular	Triangular	Triangular	Triangular	Lognormal	Lognormal

[1]Mean receptor characteristics from Richardson (1997) and CCME (2006) unless otherwise stated.
[2]Soil loadings from Kissel *et al.* (1996; 1998) as referenced in CCME (2006).
[3]Time spent outdoors by infant, toddler or child is assumed to be equivalent to that of an adult if child or infant is assumed to be accompanied by an adult.

Appendix V. Typical Values for Intakes of Air, Water, Soil, Dust and Food by the Canadian General Population[1]

Intake Rates[1]	*Statistic*	*Breastfed Infant (0 to 6 mo.)*	*Non-Breastfed Infant (0 to 6 mo.)*	*Toddler (7 mo. to 4 yr)*	*Child (5 to 11 yr)*	*Teen (12 to 19 yr)*	Adult (20+ yr)
Air Inhalation (m^3/d)	Minimum	1.1	1.1	4.6	8.3	9	9.5
	Maximum	4.4	4.4	15.6	25	28.9	33
	Mean	2.18	2.18	8.31	14.52	15.57	16.57
	Std. dev.	0.59	0.59	2.19	3.38	4.00	4.05
	Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Water Ingestion[2] (L/d)	Minimum	N/A	0.1	0.2	0.2	0.2	0.2
	Maximum	N/A	0.7	0.9	1.1	2	2.7
	Mean	N/A	0.3	0.6	0.8	1	1.5
	Std. dev.	N/A	0.2	0.4	0.4	0.6	0.8
	Distribution	N/A	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Soil Ingestion[3] (kg/d)		2.0×10^{-5}	2.0×10^{-5}	8.0×10^{-5}	2.0×10^{-5}	2.0×10^{-5}	2.0×10^{-5}
Soil Inhalation[4] (m^3/d)		1.66×10^{-9}	1.66×10^{-9}	6.32×10^{-9}	1.10×10^{-8}	1.10×10^{-8}	1.26×10^{-8}
Indoor Settled Dust Ingestion (kg/d)	Minimum	8.0×10^{-8}	8.0×10^{-8}	0.00	0.00	0.00	0.00
	Maximum	1.77×10^{-3}	1.77×10^{-3}	9.4×10^{-4}	8.33×10^{-4}	3.39×10^{-5}	6.20×10^{-5}
	Mean	3.74×10^{-5}	3.74×10^{-5}	4.06×10^{-5}	3.17×10^{-5}	2.07×10^{-6}	2.51×10^{-6}
	Std. dev.	8.33×10^{-5}	8.33×10^{-5}	5.22×10^{-5}	4.58×10^{-5}	2.32×10^{-6}	3.06×10^{-6}
	Distribution	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal
Food[5] (L/d – Breastmilk) µg Zn/kg bw/d – Food)	Minimum	0.5	541.6	467.9	325.2	202.6	140.4
	Maximum	1	893.6	624.0	452.5	304.5	219.3
	Mean	0.7	717.6	545.9	388.9	253.6	179.9
	Std. dev.	N/A	58.7	26.0	21.2	17.0	13.1
	Distribution	Triangular	Lognormal	Lognormal	Lognormal	Lognormal	Lognormal

[1] Probability distribution function curves for receptor intake rates from HC (2011a) unless otherwise stated.

[2] Breast fed infants are assumed to be exclusively breastfed for 6 months and are not given drinking water. Infants that are not breastfed are assumed to consume 0.3 L of drinking water based on HC (2004).

[3] Soil ingestion rates from CCME (2006).

[4] Soil inhalation rates based on Allan *et al.* (2008) and a PM_{10} concentration of 0.76 µg/m^3 (CCME 2006).

[5] Breastfed infants are assumed to be exclusively breast fed for 6 months and non-breastfed infants are assumed to be fed a mixture of milk, formula and table food. Applicable to infant receptors only.

Appendix VI. Typical Values for Average Body Weights and Intakes of Air, Water and Soil by the Canadian General Population used in SQG Calculation

Age (years)	Body weight[1] (kg)	Air intake[2] (m^3/d)	Water intake[1] (L/d)	Soil intake[1] (g/d)	Soil inhalation[3,4] (g/d)	Settled indoor dust ingestion[5](g/d)
0–6 months	8.2	2.2	0.3	0.02	0.0000017	0.037
7 months – 4	16.5	8.3	0.6	0.08	0.0000063	0.041
5–11	32.9	14.5	0.8	0.02	0.000011	0.032
12–19	59.7	15.6	1.0	0.02	0.000012	0.0021
20+	70.7	16.6	1.5	0.02	0.000013	0.0025

[1] HC (2010a) and CCME (2006)
[2] Allan *et al.* (2008)
[3] Health Canada (2010a)
[4] Air intake (m^3/d) x average airborne concentration of respirable particulate (0.00076 g/m^3)
[5] Wilson *et al.* (2013).

Appendix VII. Estimated Total Daily Zinc Intake by Age Class for the Canadian General Population[1]

Medium of exposure	Daily Zinc Intake (µg/kg bw/day)					
	BF-Infant (0–6 mo)	**NBF-Infant (0–6 mo)**	**Toddler (7mo– 4yr)**	**Child (5–11 yr)**	**Teen (12–19 yr)**	**Adult (20 yr+)**
AIR						
Ambient Air (Inhalation)	0.000134	0.000134	0.000249	0.000327	0.000108	0.0000958
Indoor Air (Inhalation)	0.00252	0.00252	0.00459	0.00395	0.00236	0.00215
DRINKING WATER						
Drinking Water (Ingestion)	NA	0.0784	0.0693	0.0486	0.039	0.044
INDOOR SETTLED DUST						
Settled Dust (Ingestion)	0.481	0.481	0.373	0.136	0.00576	0.00544
Settled Dust (Inhalation)	0.000000373	0.000000373	0.000000692	0.000000905	0.000000307	0.000000267
Settled Dust (Dermal)	0.144	0.144	0.103	0.0775	0.0418	0.0388
SOIL						
Soil (Ingestion)	0.0877	0.0877	0.170	0.0213	0.0117	0.00979
Soil (Dermal)	0.0201	0.0201	0.0145	0.0111	0.00588	0.00554
FOOD						
Food (Ingestion)	247	708	542	385	251	178
TOTAL						
TOTAL EDI*	253	712	545	387	251	178

[1] Median values of estimated daily intake values for each age class were modelled based on receptor characteristic details listed in Appendix IV and V and the probability distribution functions of typical concentrations of air (indoor and outdoor), drinking water, indoor settled dust, soil and food based on details in Appendix III. The probabilistic modelling of the EDI was completed as described in HC (2011a). The median value (50th percentile) was chosen to represent the EDI values for the Canadian population. Food intake EDIs were determined deterministically.

* Since a probabilistic method was used to develop the EDIs, the total EDI is not the sum of all sub-EDIs for each age catagory. The total EDI and each sub-EDI have individual probability distribution functions. The 50th percentile (median) for each distribution is displayed in the above table.

Appendix VIII. Consulted Data on the Toxicity of Zinc to Soil Microbial Processes

Microbial Process	Effect	Endpoint*	Concentration (mg/kg)	Zn Compound	Exposure Period	Soil Type	pH	OM %	Clay %	Extraction Method	Reference
Nitrification	58% reduction	EC	327‡	$ZnSO_4$	10 days	Webster loam	5.8	2.58	23	Nominal	Liang and Tabatabai 1978
Nitrification	14% reduction	EC	327‡	$ZnSO_4$	20 days	Webster loam	5.8	2.58	23	Nominal	Liang and Tabatabai 1977
	12% reduction	EC				Judson silty clay	6.6	2.95	45		
	14% reduction	EC				Okoboji silty clay loam	7.4	5.45	34		
Nitrification	no reduction	EC	1,074‡	ZnO	6 weeks	Bagshot sand	6.0	2.2	5.5	Nominal	Bhuiya and Cornfield 1974
	13% reduction	EC					7.0				
Nitrification	67% reduction	EC	100	$ZnSO_4$	2 weeks	Cecil sandy loam	6.2	1.6	7.6	Nominal	Wilson 1977
	70% reduction	EC			3 weeks						
	67% reduction	EC	100		3 weeks	Leefield loamy sand	7.4	1.14	2.4		
	31% reduction	EC			4 weeks						
	36% reduction	EC			5 weeks						
	20% reduction	EC			7 weeks						
	100% reduction	EC	1000		7 weeks	Cecil sandy loam	6.2	1.6	7.6		
	100% reduction	EC				Decatur clay loam	6.8	2.37	28.1		
	100% reduction	EC				Leefield loamy sand	7.4	1.14	2.4		
Respiration	44% reduction	EC	1,000	$ZnCl_2$	70 weeks	sand	7.0	1.6	2	Nominal	Doelman and Haanstra 1984
	40% reduction		400		43 weeks	sandy loam	6.0	5.7	9		
	38% reduction		8,000		90 weeks	silt loam	7.7	2.4	19		
Respiration	45% reduction	EC	3,270	$ZnSO_4$	45 days	Sharpsburg	8.2	4.7	11	Nominal	Lighthart *et al*. 1983
	18% reduction	EC	33								
	50% reduction	EC	3,270			Walla Walla silt loam	7.2	1.7	21		
	20% reduction	EC	327			Crider silt loam	6.7	3.1	27		
	40% reduction	EC	3,270			Toledo clay	7.0	5.5	51		
Denitrification	40% reduction	EC	250	$Zn(NO_3)_2$	21 days	silt loam	6.75	1.8	28.1	Nominal	Bollag and Barabasz 1979

Microbial Process	Effect	Endpoint*	Concentration (mg/kg)	Zn Compound	Exposure Period	Soil Type	pH	OM %	Clay %	Extraction Method	Reference
	65% reduction	EC	500								
Respiration	45% reduction	EC	100	$ZnSO_4$	8 weeks	loamy sand	4.9	2.1	5.2	Nominal	Cornfield 1977
Nitrogen mineralization	8% reduction	EC	1074‡	ZnO	6 weeks	Bagshot sand	7.0	2.2	5.5	HCl 6N	Bhuiya and Cornfield 1974
		NOEC	1074‡				6.0				
Nitrogen fixation	100 % reduction	EC	385	$ZnSO_4$	18 months	sandy loam	6.5	NR	9	aqua regia digestion	Chaudri *et al.* 1992
	90 % reduction	EC	282								
		NOEC	455		2 months						
Glucose mineralization	13% reduction	EC	100	$ZnCl_2$	24 hours	sandy clay loam	6.7	1.17	NR	Nominal	Ohya *et al.* 1985
	33% reduction	EC	300								
	44% reduction	EC	1000								
	11% reduction	EC	1000		96 hours						
Acid phosphatase activity	32% reduction	EC	1643‡	$ZnSO_4$	1.5 hours	clay loam	7.8	3.74	30	Nominal	Juma and Tabatabai 1977
	33% reduction	EC	1643‡			silty clay	7.4	5.45	34		
	30% reduction	EC	1643‡			loam	5.8	2.58	23		
Alkaline phos-phatase activity	59% reduction	EC	1643‡	$ZnSO_4$	1.5 hours	clay loam	7.8	3.74	30	Nominal	Juma and Tabatabai 1977
Urea hydrolysis		LOEC	70	$ZnCl_2$	6 weeks	sand	7.0	1.6	2	Nominal	Doelman and Haanstra 1986
	50% reduction	EC	420								
	90% reduction	EC	2490								
		LOEC	160		18 months						
	50% reduction	EC	290								
	90% reduction	EC	2490								
		LOEC	30		6 weeks	sandy loam	6.0	5.7	19		
	50% reduction	EC	480								
	90% reduction	EC	8320								
		LOEC	1		18 months						
	50% reduction	EC	110								
	90% reduction	EC	17,400								

Microbial Process	Effect	Endpoint*	Concentration (mg/kg)	Zn Compound	Exposure Period	Soil Type	pH	OM %	Clay %	Extraction Method	Reference
		LOEC	30		6 weeks	silt loam	7.7	2.4	19		
	50% reduction	EC	1,030								
	90% reduction	EC	38,200								
		LOEC	NR		18 months						
	50% reduction	EC	NR								
	90% reduction	EC	NR								
		LOEC	460		6 weeks	clay	7.5	3.2	60		
	50% reduction	EC	1,780								
	90% reduction	EC	6,820								
		LOEC	8		18 months						
	50% reduction	EC	90								
	90% reduction	EC	980								
Ammonium oxidation *Nitrosomonas communis* (strain Nm2)	Inhibition	EC_{50}	171	$ZnSO_4$	48 hours	2.2 % total carbon, 76 % sand, 21 % silt	5.3		3	nominal	Frühling *et al.* 2001
Ammonium oxidation autochthonous microflora	Inhibition	EC_{50}	118	$ZnSO_4$	24 hours						

* The EC endpoints represent the percentage of adverse effects compared to controls, as calculated by CCME from the data presented by the author(s).
NR = not reported
‡ Single concentration study

Appendix IX. Consulted Data on the Toxicity of Zinc to Terrestrial Plants

Organism	Effect (% reduction)	Endpoint *	Conc. (mg/kg)	Exposure Period	Chemical form	Soil pH	Test Substrate	Extraction Method	Reference
Black Spruce *Picea mariana*	On other foliar nutrient conc.	NOEC	1,200	field study	Zn (from smelter)	4.9	sandy loam	HF/HNO_3 /$HClO_4$	Hogan and Wotton 1984
Jack Pine *Pinus banksiana*	On other foliar nutrient conc.	NOEC	1,200						
Lettuce *Lactuca sativa*	Yield	NOEC	1,425	45 days	Zn from galvanized metal	7.1	drumlin soil, 6.2% organic matter (OM)	0.1N HCl	Jones 1982
Radish *Raphanus sativus*	Yield	NOEC	1,425						
Corn *Zea mays*	Yield	NOEC	1,425	45 days	Zn from galvanized metal	7.1	drumlin soil, 6.2% OM	0.1N HCl	Jones *et al.* 1987
Endive *Cichorium endiva*	Yield (53% reduction)	EC	60	growing season	$ZnSO_4$	4.2	sand, 4.4% OM, 3% clay	H_2SO_4 / HNO_3	Smilde *et al.* 1992
	(91% reduction)	EC	80						
Jack Pine *P. banksiana*	Shoot yield (6% reduction)	EC	50	12 weeks	$ZnCl_2$	6.0	sandy loam, 1.5% OM	nominal	Dixon and Buschena 1988
	Root yield (25% reduction	EC	25						
Beech *Fagus grandifolia*	Growth ring size (48% reduction)	EC	65.4	1years	$ZnSO_4$	4.8	mixture of sand, peat, forest soil	nominal	Hagemeyer *et al.* 1993
	Growth ring size (50% reduction)	EC	65.4	2 years					
	Mortality	LC_{100}	490	1 year					

Organism	Effect (% reduction)	Endpoint *	Conc. (mg/kg)	Exposure Period	Chemical form	Soil pH	Test Substrate	Extraction Method	Reference
Rice *Oryza sativa*	Yield	EC_{25}	30,000	15 weeks	ZnO	5.95	alluvial soil	nominal	Muramoto *et al.* 1990
		EC_{25}	50,000						
Wheat *Triticum estiva*	Yield (64% reduction)	EC	1,000	23 weeks					
	(82% reduction)	EC	10,000						
	(99% reduction)	EC	30,000						
Lettuce *L. sativa*	Seedling emergence	LOEC	410	5 days	$ZnCl_2$	4.1	artificial soil, 4.8% OM	HNO_3 + H_2O_2 + HCl	EC 1995
Lettuce *L. Sativa*	Mortality	LC_{100}	303	5 weeks	$ZnSO_4$	4.9	fine sandy loam, 1.9% OM, 16% clay	HNO_3 + $HClO_4$ + HF	MacLean 1974
Alfalfa *Medicago sativa*	Yield (71% reduction)	EC	303	16 weeks					
Corn *Z. Mays*	Mortality	LC_{100}	1,400	7 weeks	$ZnSO_4$	5.5	fine sandy loam	0.5N HCl + DTPA + $CaCl_2$	Mortvedt and Giordano 1975

* The EC endpoints represent the percentage of adverse effects compared to controls, as calculated by CCME from the data presented by the author(s).

Appendix X. Consulted Data on the Toxicity of Zinc to Terrestrial Invertebrates

Organism	Effect (% reduction)	Endpoint *	Concentration (mg/kg)	Chemical form	Exposure Period	pH	Test Substrate	Extraction Method	Reference
Earthworm *E. foetida*	Mortality	LC_{50}	80	$ZnSO_4$	30 days	7.3	clay, 8.9% OM, 46% clay	HCl + HNO_3 (ICP)	Sheppard *et al.* 1993
		LC_{50}	460			6.3	sand, 3% clay		
		LC	600			7.9	silty clay, 2.7% OM, 43% clay		
Earthworm *E. foetida*	Cocoon production	LOEC	2,000	$Zn(C_2H_3O_2)_2$	8 weeks	NR	metal mixed with horse manure over screened soil	nominal	Malecki *et al.* 1982
		LOEC	2,000	$ZnCl_2$					
		LOEC	2,000	$Zn(NO_3)_2$					
		LOEC	4,000	ZnO					
		LOEC	500	ZnS					
		LOEC	500	$ZnCO_3$					
	Body weight	LOEC	4,000	$Zn(C_2H_3O_2)_2$					
		LOEC	2,000	$ZnCl_2$					
		LOEC	2,000	$Zn(NO_3)_2$					
		LOEC	4,000	ZnO					
		LOEC	2,000	ZnS					
		LOEC	>40,000	$ZnCO_3$					
Earthworm *E. foetida*	Mortality	LC_{50}	13 $\mu g \cdot cm^{-2}$	$Zn(C_2H_3O_2)_2$	48 hours	NR	filter paper contact test	nominal	Neuhauser *et al.* 1985
		LC_{50}	10 $\mu g \cdot cm^{-2}$	$ZnCl_2$					
		LC_{50}	10 $\mu g \cdot cm^{-2}$	$Zn(NO_3)_2$					
		LC_{50}	13 $\mu g \cdot cm^{-2}$	ZnS					
Earthworm *E. foetida*	Growth	LOEC	1,300 to 13,000	$ZnSO_4$	8 weeks	6.5 to 7.0	silt loam	nominal	Hartenstein *et al.* 1981

Organism	Effect (% reduction)	Endpoint *	Concentration (mg/kg)	Chemical form	Exposure Period	pH	Test Substrate	Extraction Method	Reference
Earthworm *Eisenia Andrei*	Mortality	NOEC	320	$ZnCl_2$	3 weeks	6.0	sandy loam, 10% OM, 20% clay	HNO_3/HCl	Van Gestel *et al.* 1993
Earthworm *E. foetida*	Mortality (90%) Mortality (93%)	LC LC	900 1,000	$ZnCl_2$	14 days	4.2 4.0	artificial soil, 4.7% OM	HNO_3 + H_2O_2 + HCl	EC 1995
Wood Lice *Porcellio scaber*	Mortality	LC_{50}	1,090	$Zn(NO_3)_2$	100 days	NR	leaf litter	nominal	Hopkin and Hames 1994
Potworm *Enchytreaus doerjesi*	Population growth rate (5 worms per replicate) (10 worms per replicate) (20 worms per replicate) (40 worms per replicate) (80 worms per replicate)	EC_{50}	 2,201 3,534 7,755 5,104 4,385	$ZnCl_2$	4 weeks	6	artificial soil, 69% sand, 10% peat	nominal	Kramarz *et al.* 2005

* The EC endpoints represent the percentage of adverse effects, compared to controls as calculated by CCME from the data presented by the author(s).
NR = not reported
est. = estimated

Appendix XI. Consulted Data on the Acute and Chronic Toxicity of Zinc to Mammals

Organism	Effect (% Decrease)	Endpoint	Diet Concentration (mg/kg)	Average Dose mg/kg BW/d	Form of Zinc (Exposure Period)	Reference
Cheviot sheep	Viability of offspring Feed intake Body weight gain	NOEC	150	NR	$ZnSO_4$ (80 days)	Campbell and Mills 1979
Rats	Urine excretion (75%) Renal function Body weight gain Feed consumption Organ weights	EC NOEC		640 160	Zn acetate (3 months)	Llobet *et al.* 1988
Sheep	Feed consumption (53%) Weight loss (NQ) Feed consumption (100%) Water consumption (75%)	EC EC EC EC	4,000 6,000 	123 (calc.) 178 (calc.) 	ZnO (10 weeks) ZnO (11 days) 	Ott *et al.* 1966
Sprague-Dawley rats	Bone mineral density (6% reduction) Femur morphometry and mineralization Serum markers of bone metabolism Femur and hepatic mineral concentrations (32% lower hepatic copper concentrations)	LOAEL NOAEL NOAEL LOAEL	300 300 300 300	6.28 6.28 6.28 6.28	($ZnCO_3$) 3 weeks	Jamieson *et al.* 2006
Mice	DNA damage (represented as comet tail length)	LOAEL LOAEL LOAEL LOAEL		5.7 mg/kg BW (24 h) 5.7 mg/kg BW (48 h) 5.7 mg/kg BW (72 h) 8.55 mg/kg BW (96 h) (no significant effects observed at 1 wk)	$ZnSO_4$ (1 exposure event and experiment ran for 1 week)	Banu *et al.* 2001

NR = not reported
NQ = not quantified
calc. = calculated from data reported by the author(s)

Appendix XII. Consulted Data on the Acute and Chronic Toxicity of Zinc to Birds

Organism	Effect (% Decrease)	Endpoint	Diet Concentration (mg/kg)	Average Dose mg/kg BW·d	Form of Zinc (Exposure Period)	Reference
Mallard duck	Body weight gain	NOEC	100	9.32	Zn metal shot (28 days)	French *et al.* 1987
	Body weight gain	NOEC	150	14.4		
Mallard duck	Mortality (60%)	LC	3,000	109 (calc.)	$ZnCO_3$ (60 days)	Gasaway and Buss 1972
	Mortality (100%)		6,000	158 (calc.)	$ZnCO_3$ (40 days)	
Poultry	Food consumption (11%)	EC	2,000	129.4 (calc.)	ZnO (28 days)	Dewar *et al.* 1983
	Development of pancreatic lesions (62%)					
	Body weight (54%)	EC	4,000	494.3 (calc.)		
	Food consumption (17%)					
	Development of pancreatic lesions (100%)					
Poultry (boiler chicks)	Lymphoid cell degeneration	effects observed (NQ)	1,000		8 weeks	Donmez *et al.* 2003

NQ = not quantified
calc. = calculated from data reported by the author(s)

Appendix XIII. Selected Microbial Toxicological Studies for Zinc

Species\Process	Effect (% Decrease)	Endpoint *	Concentration (mg/kg)	Form of Zn (Exposure Period)	Soil pH	Test Substrate	Extraction Method	Reference
Nitrification	Inhibition (24%)	EC*	327	$ZnSO_4$ (10 days)	7.8	3.74% OC 30% clay	nominal	Liang and Tabatabai 1978
	Inhibition (39%)	EC*	327		7.4	5.45 % OC 34% clay		
Nitrification	Inhibition (15%)	EC*	327	$ZnSO_4$ (20 days)	7.8	3.74% OC 30% clay	nominal	Liang and Tabatabai 1977
N-Mineralization	Inhibition (32%)	EC*	1,074	ZnO (6 weeks)	7.7	2.2% OM	6N HCl	Bhuiya and Cornfield 1974
Nitrification	Inhibition (33%)	EC*	1,074			5.5% clay		
Respiration CO_2 release	Reduction (21%)	EC*	10	$ZnSO_4$ (8 weeks)	4.9	2.1% OM, 5.2% clay	nominal	Cornfield 1977
	Reduction (20%)	EC*	10	$ZnSO_4$ (2 weeks)				
	Reduction (24%)	EC*	100					
Respiration CO_2 release	Reduction (32%)	EC*	33	$ZnSO_4$ (45 days)	8.2	4.7% OM 11% clay	nominal	Lighthart *et al.* 1983
	Reduction (20%)	EC*	327					
	Reduction (20%)	EC*	327		7.2	1.7% OM 21% clay		
	Reduction (25%)	EC*	327		6.7	3.1% OM 27% clay		
	Reduction (20%)	EC*	327		7.0	5.5% OM 51% clay		
	Reduction (30%)	EC*	3,270		6.7	3.1% OM 27% clay		
Respiration CO_2 release	Reduction (16%)	EC*	1,074	ZnO (12 weeks)	6.0	2.2% OM, 5.5% clay	6N HCl	Bhuiya and Cornfield 1972
Respiration CO_2 release	Reduction (26%)	EC*	3,000	$ZnCl_2$ (82 weeks)	4.4	12.8% OM	nominal	Doelman and Haanstra 1984
						5% clay		
	Reduction (26%)		3,000	$ZnCl_2$ (80 weeks)	7.5	3.2% OM 60% clay		
Carbon mineralization	Reduction	NOEC LOEC	1,000 5,000	$ZnSO_4$ (14 days)	5	5.8% OM, 9% clay	nominal	Bewley and Stotzky 1983
Respiration CO_2 release	Reduction	NOEC LOEC	300 620	$ZnSO_4 \cdot 7H_2O$ (52 days)	5	5.0% OM 1.6% clay	Aqua Regia	Lahr *et al.* 2008

Species\Process	Effect (% Decrease)	Endpoint *	Concentration (mg/kg)	Form of Zn (Exposure Period)	Soil pH	Test Substrate	Extraction Method	Reference
Nitrification	Inhibition	EC_{50}	240	$ZnCl_2$ (35 days)	4.89	sandy loam, CEC 1.79 cmol(+)/kg	nominal	Cela and Sumner 2002
	Inhibition	EC_{50}	440	$ZnCl_2$ (35 days)	6.88	sandy loam, CEC 1.79 cmol(+)/kg		
	Inhibition	EC_{50}	200	$ZnCl_2$ (35 days)	7.23	sandy, CEC 0.20 cmol(+)/kg		
Nitrification	Inhibition	EC_{50}	559	$ZnCl_2$ (21 days)	5.6–6.4	spiked uncontaminated sandy loam	Aqua Regia	Mertens *et al.* 2006
	Inhibition	EC_{50}	1,908	$ZnCl_2$ (21 days)	5.6–6.4	spiked long-tern Zn-contaminated sandy loam		
Substrate–induced nitrification (SIN)	Reduction	EC_{50}	687 (median of 12 soils)	$ZnSO_4$ (28 days)	4.0–7.6	0.9–5.6% OC 4–66% clay 3–55 CEC ($cmol_c$/kg) 5–80 Zn background (mg/kg)	Aqua Regia	Broos *et al.* 2007
Substrate–induced respiration (SIR)	Reduction	EC_{50}	2,377 (median of 8 soils)		4.4–7.6	0.9–3.4% OC 4–66% clay 3–55 CEC ($cmol_c$/kg) 5–80 Zn background (mg/kg)		
Nitrification	Decrease	EC_{50}	210	$ZnSO_4$ (28 days)	9.0	Sterilized soil reinoculated with 17 month incubated clean soil	nominal	Rusk *et al.* 2004

Species\Process	Effect (% Decrease)	Endpoint *	Concentration (mg/kg)	Form of Zn (Exposure Period)	Soil pH	Test Substrate	Extraction Method	Reference
						4 % clay 0.55% total carbon 6.4 CEC ($cmol^+/kg$)		
	Decrease	EC_{50}	850			Sterilized soil reinoculated with 17 month incubated Zn-exposed (670–890 mg/kg) soil 4% clay 0.55% total carbon 6.4 CEC ($cmol^+/kg$)		

* The EC endpoints represent the effects concentration as calculated by CCME from the data presented by the author(s).

Appendix XIV. Selected Plant and Invertebrate Toxicological Studies for Zinc

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
Onion *Allium cepa*	Dry matter yield (18% reduction)	LOEC	400	$ZnSO_4$ (8 weeks)	8.3	clay loam, 0.28% OM, 24% clay	nominal	Dang *et al.* 1990
Jack Pine *Pinus banksiana*	Root yield (36% reduction)	EC	50	$ZnCl_2$ (12 weeks)	6.0	sandy loam, 1.5% OM	nominal	Dixon and Buschena 1988
White Spruce *Picea glauca*	Shoot yield (13% reduction)	EC	50					
	Root yield (28%) reduction	EC	50					
Radish *Raphanus sativus*	Seedling emergence	NOEC	100	$ZnCL_2$ (3 days)	4.1	artificial soil 4.8% OM	$HNO_3 + H_2O_2$ + HCl	EC 1995
		LC_{25}	160					
	(37% reduction)	LOEC	200					
		LC_{50}	280					
		NOEC	230		4.2	artificial Soil 4.7% OM		
		LC_{25}	420					
	(34% reduction)	LOEC	490					
		LC_{50}	670					
	(11% reduction)	NOEC	130		4.0	artificial soil 6.3% OM		
		LOEC	240					
		LC_{25}	320					
		LC_{50}	520					
Lettuce *Lactuca sativa*	Seedling emergence	NOEC	220	$ZnCl_2$ (5 days)	4.2	artificial soil 4.7% OM	$HNO_3 + H_2O_2$ + HCl	EC 1995
		LC_{25}	350					
	(49% reduction)	LOEC	490					
		LC_{25}	500					
		NOEC	250		4.0	artificial soil 6.3% OM		
		LC_{25}	470					
		LC_{50}	720					
		NOEC	200		4.1	artificial soil 10.4% OM		
		LC_{25}	280					
		LC_{50}	400					
Earthworm *Eisenia foetida*	Mortality	NOEC	500	$ZnCl_2$ (14 days)	4.2	artificial soil 4.7% OM	$HNO_3 + H_2O_2$ + HCl	EC 1995
		LC_{25}	700					
		LC_{50}	800					
		NOEC	400		4.0	artificial soil		

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
		LC_{25} LC_{50}	500 700			6.3% OM		
	(40% mortality)	NOEC LC_{25} LOEC LC_{50}	300 500 600 700		4.1	artificial soil 10.4% OM		
Beech *Fagus grandifolia*	Shoot growth (21% reduction) Shoot growth (39% reduction)	EC EC	65.4 65.4	$ZnSO_4$ (1 year) (2 years)	4.8	mix: sand/peat/forest soil	nominal	Hagemeyer *et al.* 1993
Blackgram *Vigna mungo*	Yield (22% reduction) Yield (45% reduction)	EC EC	200 250	$ZnSO_4$ (65 days)	6.2	NR	nominal	Kalyanaraman and Sivagurunathan 1994
Corn *Zea mays*	Yield (13% reduction)	EC	303	$ZnSO_4$ (6 weeks)	4.9	fine sandy loam 16% clay 1.9% OM	HNO_3 + $HClO_4$ + HF	MacLean 1974
		NOEC	329		7.5	sandy loam 16% clay 2.4% OM		
		NOEC	328		7.2	sandy loam 13.3% clay 5.6% OM		
Lettuce *L. Sativa*	Dry matter yield	NOEC	329	$ZnSO_4$ (5 weeks)	7.5	sandy loam 16% clay 2.4% OM		
		NOEC	328		7.2	sandy loam 13.3% clay 5.6% OM		
Alfalfa *Medicago sativa*	Dry matter yield	NOEC	329	$ZnSO_4$ (16 weeks)	7.5	sandy loam. 16% clay 2.4% OM		
		NOEC	328		7.2	sandy loam 13.3% clay		
						5.6% OM		
Corn *Zea mays*	Yield	EC_{50}	240	$ZnSO_4$ (7 weeks)	5.5	sandy loam	nominal	Mortvedt and Giordano 1975
Rice *Oryza sativa*	Yield (23% reduction)	EC	10,000	ZnO (15 weeks)	5.95	alluvial soil	nominal	Muramoto *et al.* 1990

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
Earthworm *E. Foetida*	Mortality	LC_{50}	662	$Zn(NO_3)_2$ (14 days)	6.0	artificial sandy loam 10% OM, 20% clay	nominal	Neuhauser *et al.* 1985
Turnip *Brassica rapa*	First bloom	EC_{50}	600	$ZnSO_4$	7.2	clay 23% OM 48% clay	$HCl + HNO_3$ (ICP)	Sheppard *et al.* 1993
	Seed yield	EC_{50}	700					
	Seedling emergence	NOEC	1,000					
	First bloom	EC_{50}	25		6.3	sand, negligible OM,3% clay		
	Seed yield	EC_{50}	25					
	Seedling emergence	EC_{50}	65					
	First bloom	EC_{50}	650		7.9	silty clay 2.7% OM 43% clay		
	Seed yield	EC_{50}	700					
	Seedling emergence	EC_{50}	650					
Lettuce *L. Sativa*	Seedling emergence	NOEC	1,000	$ZnSO_4$	7.2	clay 23% OM 48% clay	$HCl + HNO_3$ (ICP)	Sheppard *et al.* 1993
		EC_{50}	200		6.3	sand negl. OM 3% clay		
		NOEC	1,000		7.9	silty clay 2.7% OM 43% clay		
Earthworm *Lumbricus terrestris*	Mortality	LC_{50}	80	$ZnSO_4$ (30 days)	7.2	clay, 23% OM, 48% clay	$HCl + HNO_3$ (ICP)	Sheppard *et al.* 1993
		LC_{50}	600		6.3	sand, negl. OM, 3% clay		
		LC_{50}	600		7.9	silty clay, 2.7% OM, 43% clay		
Spinach *Spinacia oleracea*	Yield (27% reduction)	EC NOEC	80 20	$ZnSO_4$ (growing season)	4.2	sand, 4.4% OM, 3% clay	H_2SO_4 / HNO_3	Smilde *et al.* 1992
		NOEC	160		7.2	loam, 3.7% OM, 40% clay		
Earthworm	Mortality	LC_{50} LC_{50}	1,010 745	$Zn(NO_3)_2$ (56 days)	6.3	artificial sandy loam 10% OM,	HNO_3	Spurgeon *et al.* 1994

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
Eisenia fetida		NOEC	289 (est.)			20% clay		
	Cocoon production	EC_{50}	276					
		NOEC	199 (est.)					
Earthworm *Eisenia fetida*	Cocoon prodction	EC_{50}	136	$Zn(NO_3)_2$ (21 days)	6	artificial (OECD) 5% OM, 20% clay	Nitric acid	Spurgeon and Hopkin 1996
		EC_{50}	462			artificial (OECD) 10% OM, 20% clay		
		EC_{50}	592			artificial (OECD) 15% OM, 20% clay		
Earthworm *E. Andrei*	Cocoon production	NOEC	320	$ZnCl_2$ (3 weeks)	6.0	artificial sandy loam, 10% OM, 20% clay	HNO_3 / HCl	Van Gestel *et al.* 1993
	Cocoon production (31% reduction)	LOEC	560					
	Cocoon production	EC_{50}	659					
	Number of juveniles produced/worm	EC_{50}	512					
	Body weight gain	NOEC	560					
Lettuce *Lactuca sativa*	Growth	EC_{50}	276	$Zn(NO_3)_2$ (31 days)	4.7	silty clay, 5.66% total carbon	Aqua regia	Stevens *et al.* 2003
	Growth	EC_{50}	10		4.8	sand, 0.31% total carbon		
	Growth	EC_{50}	383		7.8	sand, 0.55% total carbon		
	Growth	EC_{50}	328		6.5	sandy loam, 0.96% total carbon		
	Growth	EC_{50}	274		5.1	sandy clay loam, 2.83% total carbon		
Wheat *Triticum aestivum L.*	Growth	EC_{20}	980	$ZnSO_4$ (21 days)	7.6	12.4% clay, 1.1% OC	HNO_3+HCI (3:1) reverse aqua regia	Warne *et al.* 2008
	Growth	EC_{20}	275		5.6	3.9% clay, 0.9% OC		
	Growth	EC_{20}	300		4.5	15.7% clay, 1.4% OC		
	Growth	EC_{20}	6,140		7.9	65.5% clay, 1.4% OC		
	Growth	EC_{20}	880		6.0	69.0% clay, 2.9% OC		
	Growth	EC_{20}	1,100		5.0	22.9% clay, 2.0% OC		
	Growth	EC_{20}	1,000		4.0	5.4% clay, 5.7% OC		

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
	Growth	EC_{20}	710		5.0	25.0% clay, OC not available		
	Growth	EC_{20}	400		4.4	17.3% clay, 1.3% OC		
	Growth	EC_{20}	685		5.4	40.9% clay, 1.8% OC		
	Growth	EC_{20}	825		4.9	23.5% clay, 3.5% OC		
	Growth	EC_{20}	860		6.3	26.9% clay, 1.9% OC		
	Growth	EC_{20}	615		6.7	10.4% clay, 1.8% OC		
	Growth	EC_{20}	460		4.6	5.9% clay, 2.6% OC		
Chinese cabbage *Brassica Chinensis L.*	Shoot dry matter yield	EC_{10}	244	$ZnSO_4 \cdot 7H_20$ (35 days)	7.15	loamy soil, 3.87% OC		Long *et al.* 2003
Pak choi *Brassica Chinensis L.*	Shoot dry matter yield	EC_{10}	277					
Celery *Apium graveolens L.*	Shoot dry matter yield	EC_{10}	204					
White clover *Trifolium repens*	Biomass (36% reduction)	LOEC	460	$ZnSO_4$ (56 days)	6.5–7.7	5-year aged flood plain sediment	nominal	Bernhard *et. al*. 2005
Rapeseed *Brassica napus*	Biomass (53% reduction)	LOEC	460					
Creeping bent grass *Argostis stolonifera*	Biomass (19% reduction)	LOEC	460					
Barley *Hordeum vulgare*	Root dry weight	EC_{25}	229	$ZnSO_4 \cdot 7H_20$ (45 days)	7.8	sandy loam	nominal	Aery and Jagetiya 1997
Jack bean	Shoot dry weight	EC_{25}	298	$ZnSO_4$ (11 days)	5.5	loamy soil	nominal	Andrade *et al*. 2009
Barley	Growth	EC_{50}	No fit	$ZnCl_2$ (14 days)	6.7	field soil	Aqua regia	Hamels *et al*. 2014
	Growth	EC_{50}	2,040	$ZnCl_2$ (14 days)	6.7	spiked soil 2% OC		
	Growth	EC_{50}	3,600	$ZnCl_2$ (14 days)	7.1	field soil		
	Growth	EC_{50}	1,750	$ZnCl_2$ (14 days)	7.1	spiked soil 1% OC		

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
	Growth	EC_{50}	5,200	$ZnCl_2$ (14 days)	6.1	field soil		
	Growth	EC_{50}	1,870	$ZnCl_2$ (14 days)	6.1	spiked soil 5% OC		
	Growth	EC_{50}	240	$ZnCl_2$ (14 days)	6.6	Field soil		
	Growth	EC_{50}	140	$ZnCl_2$ (14 days)	6.6	spiked soil 5% OC		
	Growth	EC_{50}	4,080	$ZnCl_2$ (14 days)	5.8	Field soil		
	Growth	EC_{50}	920	$ZnCl_2$ (14 days)	5.8	spiked soil 5% OC		
	Growth	EC_{50}	7,300	$ZnCl_2$ (14 days)	6.2	field soil		
	Growth	EC_{50}	2,170	$ZnCl_2$ (14 days)	6.2	spiked soil 23% OC		
	Growth	EC_{50}	6,100	$ZnCl_2$ (14 days)	5.3	field soil		
	Growth	EC_{50}	790	$ZnCl_2$ (14 days)	5.3	spiked soil 1% OC		
	Growth	EC_{50}	1,260	$ZnCl_2$ (14 days)	6.1	field soil		
	Growth	EC_{50}	230	$ZnCl_2$ (14 days)	6.1	spiked soil 7% OC		
	Growth	EC_{50}	1,490	$ZnCl_2$ (14 days)	7.6	field soil		
	Growth	EC_{50}	450	$ZnCl_2$ (14 days)	7.6	spiked soil 14% OC		
Compost worm *Eisenia fetida*	Reproduction	EC_{50}	705	$ZnCl_2$ (21 days)	6	artificial soil (OECD)	$HCl:HNO_3$ 1:5	Lock and Janssen 2003b
	Reproduction	EC_{50}	764	ZnO (21 days)	6	artificial soil (OECD)		
	Reproduction	EC_{50}	587	Zn fine powder (21 days)	6	artificial soil (OECD)		
Potworm *Enchytraeus albidus*	Reproduction	EC_{50}	271	$ZnCl_2$ (42 days)	6	artificial soil (OECD)		
	Reproduction	EC_{50}	461	ZnO (42 days)	6	artificial soil (OECD)		
	Reproduction	EC_{50}	302	Zn fine powder (42 days)	6	artificial soil (OECD)		
Springtail *Folsomia candida*	Reproduction	EC_{50}	391	$ZnCl_2$ (28 days)	6	artificial soil (OECD)		
	Reproduction	EC_{50}	461	ZnO (28 days)	6	artificial soil (OECD)		

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
	Reproduction	EC_{50}	393	Zn fine powder (28 days)	6	artificial soil (OECD)		
Earthworm *Eisenia fetida*	Reproduction	EC_{50}	704	$ZnCl_2$ (21 days)	6	artificial soil (OECD)	$HCl:HNO_3$ 1:5	Lock and Janssen 2001
	Reproduction	EC_{50}	294	$ZnCl_2$ (21 days)	6.3	loamy soil (17% clay, 1.5% OM)		
Potworm *Enchytraeus albidus*	Reproduction	EC_{50}	267	$ZnCl_2$ (42 days)	6	artificial soil (OECD)		
	Mortality	LC_{50}	147	$ZnCl_2$ (14 days)	4.5	sandy soil (1% clay, 4.8% OM)		
	Reproduction	EC_{50}	92	$ZnCl_2$ (42 days)	6.3	loamy soil (17% clay, 1.5% OM)		
Springtail *Folsomia candida*	Reproduction	EC_{50}	375	$ZnCl_2$ (28 days)	6	artificial soil (OECD)		
	Reproduction	EC_{50}	78	$ZnCl_2$ (28 days)	4.5	sandy soil (1% clay, 4.8% OM)		
	Reproduction	EC_{50}	522	$ZnCl_2$ (28 days)	6.3	loamy soil (17% clay, 1.5% OM)		
Roundworm *Caenorhabditis elegans*	Mortality	LC_{50}	1,915	$Zn(NO_3)_2$ (24 hours)	4	artificial soil (10% peat moss, 20% clay, 70% sand) (ASTM)	nominal	Peredney and Williams 2000
Springtail *Folsomia candida*	Reproduction	EC_{50}	683	$ZnCl_2$ (42 days)	6	artificial soil (OECD)	$HCl:HNO_3$ 1:4	Van Gestel and Hensbergen 1997
Earthworm *Eisenia fetida*	Reproduction	EC_{50}	1,898	$Zn(NO_3)_2 \cdot 6H_20$ (42 days)	6.35	sandy loam (9.7% clay, 2.35% OM)	10 ml HNO_3 diluted to 100 ml with double dist. H_2O	Spurgeon *et al*. 2000
Red worm *Lumbricus terrestris*	Reproduction	EC_{50}	1,029	$Zn(NO_3)_2 \cdot 6H_20$ (42 days)	6.35	sandy loam (9.7% clay, 74% sand, 2.35% OM)		
Reddish brown worm *Lumbricus rubellus*	Reproduction	EC_{50}	599	$Zn(NO_3)_2 \cdot 6H_20$ (42 days)	6.35	sandy loam (9.7% clay, 74% sand, 2.35% OM)		
Grey worm *Aporrectodea caliginosa*	Reproduction	EC_{50}	442	$Zn(NO_3)_2 \cdot 6H_20$ (42 days)	6.35	sandy loam (9.7% clay, 74% sand, 2.35% OM)		

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
Springtail *Folsomia candida*	Reproduction	EC_{50}	1,749	$ZnCl_2$ (35 days)	6.4±0.04 and 7.1±0.1	Aged natural soil	5ml conc. HNO_3	Smit *et al*. 1997
Nematode Filenchus	Population size reduction	EC_{50}	141	$ZnSO_4$ (7–14 days)	4.1	sandy loam (85% sand, 11% silt, 4% clay, 1.9% OC, 3.6 CEC [$cmol_c$/kg])	HNO_3 : H_2SO_4 1:1	Korthals *et al*. 1996
Nematode Pratylenchus	Population size reduction	EC_{50}	902					
Nematode Tylenchorhynchus	Population size reduction	EC_{50}	710					
Nematode Dauer-larvae	Population size reduction	EC_{50}	1,538					
Nematode Rhabditidae	Population size reduction	EC_{50}	444					
Nematode Acrobeloides	Population size reduction	EC_{50}	493					
Nematode Eucephalobus	Population size reduction	EC_{50}	300					
Nematode Plectus	Population size reduction	EC_{50}	52					
Nematode Aphelenchoides	Population size reduction	EC_{50}	527					
Nematode Pseudhalenchus	Population size reduction	EC_{50}	1,600	$ZnSO_4$ (14 days)				
Nematode Clarkus	Population size reduction	EC_{50}	100	$ZnSO_4$ (14 days)				
Nematode Aporcelaimellus	Population size reduction	EC_{50}	145	$ZnSO_4$ (14 days)				
Earthworm *Eisenia fetida*	Reproduction	NOEC	250	$ZnCl_2$ (28 days)	6.0±0.05	OECD artificial soil adjusted to 5% clay	Nitric acid extaction	Owojori *et al.* 2009
	Reproduction (20% reduction)	LOEC	500	$ZnCl_2$ (28 days)	6.0±0.05	OECD artificial soil adjusted to 5% clay		

Organism	Effect (% decrease)	Endpoint*	Concentration (mg/kg)	Form of Zn (exposure period)	Soil pH	Test Substrate	Extraction Method	Reference
	Reproduction	NOEC	250	$ZnCl_2$ (28 days)	6.0±0.05	OECD artificial soil adjusted to 20% clay		
	Reproduction (12% reduction)	LOEC	500	$ZnCl_2$ (28 days)	6.0±0.05	OECD artificial soil adjusted to 20% clay		
Springtail *Proisotoma minuta*	Reproduction	EC50	283	$Zn(NO_3)_2 \cdot 6H_20$ (42 days)	4.88	1.32 % OC, 75% sand, 13.3 % silt, 10% clay, 36.5 CEC ($mmol_c$/kg)	nominal	Nurista *et al.* 2005

* The EC endpoints represent the effects concentration as calculated by CCME from the data presented by the author(s).
NR = not reported
negl. = negligible
est. = estimated

Appendix XV. Selected Livestock and Wildlife Toxicological Studies for Zinc

Organism	Effect (% decrease)	Endpoint *	Diet Concentration (mg/kg)	Average Dose mg/kg BW/d^{-}	Form of Zinc (exposure period)	Reference
Cheviot sheep	Number of viable offspring (64%) Feed consumption (24%) Body weight gain during pregnancy (67%)	EC	750	20 for 10 days 10 for final 10 weeks	$ZnSO_4$ (80 days)	Campbell and Mills 1979
Sheep	Body weight gain (33%) Feed consumption (15%)	EC	2,000	76.7 (calc.)	ZnO (10 weeks)	Ott *et al.* 1966
	Body weight gain (16%)	LOAEL	1,000	42.4 (calc.)		
	Feed consumption (13%)	LOAEL	1,500	57.2 (calc.)		
Sheep	Body weight gain (43%) Enlarged & pale kidneys (NQ) Decreased liver copper content (NQ)	EC	1,34.3	33.6 (calc.)	ZnO (33 days)	Davies *et al.* 1977
Poultry	Body weight (35%)	EC	5,280	1,074 (calc.)	ZnO (28 days)	Dean *et al.* 1991
Poultry	Development of pancreatic lesions (38%) Increased zinc liver concentration (NQ)	LOAEL	1,000	65.7 (calc.)	ZnO (28 days)	Dewar *et al.* 1983
Rats	Urine excretion (76%)	EC	NR	320 †	Zn acetate (3 months)	Llobet *et al.* 1988
Sprague-Dawley rats	Pups per litter (12% decrease) Live pups per litter (13% decrease) F1 male time to incisor eruption (6.4% decrease) F1 female time to incisor eruption (6.8% decrease)	LOAEL LOAEL LOAEL LOAEL		3.6 (calc.) 3.6 (calc.) 14.4 (calc.) 3.6 (calc.)	$ZnCl_2$ (170 days) $ZnCl_2$ (170 days) $ZnCl_2$ (in utero; day 15) $ZnCl_2$ (in utero; day 15)	Johnson *et al.* 2011

* The EC endpoints represent the effects concentration as calculated by CCME from the data presented by the author(s).
calc. = calculated from data reported by the author(s).
† = as reported by the author(s)

Appendix XVI. Data on the Accumulation of Zinc in Terrestrial Plant Tissues

Species	Tissue Type	pH	Soil Type	n	Zinc in Tissue (mg/kg dw)	Zinc in Soil (mg/kg dw)	BCF†	Log (BCF+1)‡	Reference
GRAMINAEAE FAMILY									
Corn *(Zea mays)*	Shoot	7.1	sand–loam	5	484	1,425	0.34	0.13	Jones *et al.* 1987
	Root	7.1		5	1,330	1,425	0.93	0.29	
	Shoot	7.2		5	25.3	67.3	0.38	0.14	
	Root	7.2		5	21	67.3	0.31	0.12	
	Root	5.8	sandy loam	4	34.6	49.7	0.7	0.23	Petruzzelli *et al.* 1989
Ryegrass *(Lolium perenne, cv Melle)*	Shoot	5.1	loam	4	450	970	0.46	0.17	Smith 1994
		4.4		5	630	1,473	0.43	0.15	
		5.3		5	370	1,473	0.25	0.1	
		5.9		5	270	1,473	0.18	0.073	
		6		5	260	1,473	0.18	0.071	
		6.2		5	250	1,473	0.17	0.068	
		6.8		5	195	1,473	0.13	0.054	
		5.9		4	320	970	0.33	0.12	
		6.2		4	330	970	0.34	0.13	
		6.5		4	280	970	0.29	0.11	
		6.6		4	285	970	0.29	0.11	
		6.9		4	230	970	0.24	0.09	
Barley *(Hordeum vulgare)*	Leaves	7.9	very fine sandy loam	4	23.2	63.6	0.36	0.13	Viets *et al.* 1954
		8.2		4	20.3	67.6	0.3	0.11	
	Shoot	6.5	loam	4	350	136	2.6	0.56	Hilber *et al.* 2007 (tissue data estimated from figures)
		6.5	loam – fresh cont.	4	2,300	2,250	1	0.3	
		6.5	loam – cont. for 4 y	4	3,000	1,811	1.7	0.43	
	Root	6.5	loam	1	50	136	0.37	0.14	
		6.5	loam – fresh cont.	1	2,100	2,250	0.93	0.29	
		6.5	loam – cont. for 4 y	1	2,200	1,811	1.2	0.34	
	Grain	7	N.R.	4	29	24.7	1.2	0.34	Dudka *et al.* 1996
		7.2		4	43	465	0.092	0.038	
		6.9		4	54	1,030	0.052	0.022	
		7.4		4	57	5,900	0.01	0.0042	
		7.4		4	58	11,375	0.005	0.0022	

Species	Tissue Type	pH	Soil Type	n	Zinc in Tissue (mg/kg dw)	Zinc in Soil (mg/kg dw)	BCF†	Log (BCF+1)‡	Reference
	Shoot	7		4	25	24.7	1	0.3	
		7.2		4	33	465	0.071	0.03	
		6.9		4	53	1,030	0.051	0.022	
		7.4		4	66	5,900	0.011	0.0048	
		7.4		4	99	11,375	0.009	0.0038	
Wheat *(Triticum aestivum)*	Leaves	7.9	very fine sandy loam	4	14.1	63.6	0.22	0.086	Viets *et al.* 1954
		8.2		4	15	67.6	0.22	0.086	
Oats *(Avena sativa)*	Leaves	7.9	very fine sandy loam	4	10.5	63.6	0.17	0.068	Viets *et al.* 1954
		8.2		4	12.1	67.6	0.18	0.072	
POACEAE FAMILY									
Tor grass (*Brachypodium pinnatum*)	Leaves	6.7	loamy sand	2	108	8,655	0.012	0.0052	Krpata *et al.* 2008
		7.1		3	132	8,177	0.016	0.0069	
		7.28		3	106.7	11,239	0.01	0.0041	
Tall oat grass *(Arrhenatherum elatius)*	Shoot	6.2	N.R.	36	948	7,744	0.12	0.049	Deram *et al.* 2006
	Root	6.2		36	3,378	7,744	0.44	0.16	
	Shoot	6.9		36	1,221	19,928	0.061	0.026	
	Root	6.9		36	3,723	19,928	0.19	0.076	
	Shoot	8.2		24	1,171	5,127	0.23	0.09	
	Root	8.2		24	2,790	5,127	0.54	0.19	
Grass (not specified)	Shoot	7	N.R.	4	26	24.7	1.1	0.32	Dudka *et al.* 1996
		7.2		4	55	465	0.12	0.049	
		6.9		4	108	1,030	0.1	0.041	
		7.4		4	141	5,900	0.024	0.01	
		7.4		4	159	11,375	0.014	0.006	
CRUCIFERAE FAMILY									
Radish *(Raphanus sativus)*	Shoot	6	N.R.	46	209	547	0.38	0.14	Davies 1992
	Root	6		46	139	547	0.071	0.03	
	Leaves	7.1	garden soil	5	549	1,425	0.38	0.14	Jones 1982
		7.1		5	49	143	0.34	0.13	
		7.1		5	42	68	0.62	0.21	
	Roots	7.1		5	167	1,425	0.12	0.049	
		7.1		5	37	143	0.26	0.1	
		7.1		5	27	68	0.4	0.15	
LEGUMINEAE FAMILY									
Soybean	Shoot	6.4	silt loam	4	1,090	1,165	0.94	0.29	Pierzynski and

Species	Tissue Type	pH	Soil Type	n	Zinc in Tissue (mg/kg dw)	Zinc in Soil (mg/kg dw)	BCF†	Log (BCF+1)‡	Reference
(Glycine max)		6.3		4	923	933	0.99	0.3	Schwab 1993
		6.4		4	725	1076	0.67	0.22	
		6.3		4	768	933	0.82	0.26	
		6.4		4	965	1076	0.9	0.28	
	Roots	6.4		4	1,248	1076	1.16	0.33	
	Leaves	7.9	very fine sandy loam	4	19.1	63.6	0.3	0.11	Viets *et al*. 1954
		8.2		4	16.4	67.6	0.24	0.093	
Lima beans (*Phaseolus vulgaris)*	Leaves	7.9	very fine sandy loam	4	18.3	63.6	0.29	0.11	Viets *et al*. 1954
		8.2		4	15.2	67.6	0.22	0.086	
Safflower (*Carthamus tinctorius)*	Leaves	7.9	very fine sandy loam	4	14.1	63.6	0.22	0.086	Viets *et al*. 1954
		8.2		4	9.3	67.6	0.14	0.057	
Alfalfa (*Medicago sativa)*	Whole shoot	7.9	very fine sandy loam	4	16.8	63.6	0.26	0.1	Viets *et al*. 1954
		8.2		4	15.6	67.6	0.23	0.09	
Red Clover (*Trifolium pratense)*	Whole shoot	7.9	very fine sandy loam	4	13.4	63.6	0.21	0.083	Viets *et al*. 1954
		8.2		4	12	67.6	0.18	0.072	
		7	N.R.	4	31	24.7	1.3	0.36	Dudka *et al* 1996
		7.2		4	54	465	0.12	0.049	
		6.9		4	93	1,030	0.09	0.037	
		7.4		4	105	5,900	0.018	0.0077	
		7.4		4	113	11,375	0.01	0.0043	
FABACEAE FAMILY									
Pea *(Pisum sativum)*	Shoots	6.5	loam	4	450	136	3.3	0.63	Hilber *et al.* 2007 (tissue concentrations estimated from figure)
		6.5	loam – fresh cont.	4	600	2,250	0.27	0.1	
		6.5	loam – cont. for 4 y	4	500	1,811	0.28	0.11	
	Roots	6.5	loam	1	250	136	1.8	0.45	
		6.5	loam – fresh cont.	1	3,400	2,250	1.5	0.4	
		6.5	loam – cont. for 4 y	1	5,000	1,811	2.8	0.58	
FABACEAE FAMILY									
Sunflower *(Helianthus annus)*	Shoots	6.5	loam	4	400	136	2.9	0.59	Hilber *et al.* 2007 (tissue concentrations
		6.5	loam – fresh cont.	4	1,000	2,250	0.44	0.16	

Species	Tissue Type	pH	Soil Type	n	Zinc in Tissue (mg/kg dw)	Zinc in Soil (mg/kg dw)	BCF†	Log (BCF+1)‡	Reference
		6.5	loam – cont. for 4 y	4	850	1,811	0.47	0.17	estimated from figure)
	Roots	6.5	loam	1	100	136	0.74	0.24	
		6.5	loam – fresh cont.	1	2,500	2,250	1.1	0.32	
		6.5	loam – cont. for 4 y	1	3,000	1,811	1.7	0.43	
COMPOSITAE FAMILY									
Lettuce *(Lactuca sativa)*	Leaves	7.1	garden soil	5	272	1,425	0.19	0.076	Jones 1982
		7.1		5	64	143	0.45	0.16	
		7.1		5	56	68	0.82	0.26	
	Roots	7.1		5	673	1,425	0.47	0.17	
		7.1		5	42	143	0.29	0.11	
		7.1		5	41	68	0.6	0.2	
CHENOPODIACEA FAMILY									
Sugarbeet (*Beta vulgaris)*	Leaves	7.9	very fine sandy loam	4	19.2	63.6	0.3	0.11	Viets *et al.* 1954
		8.2		4	22.5	67.6	0.33	0.12	
SOLANACEA FAMILY									
Potato *(Solanum tuberosum)*	Leaves	7.9	very fine sandy loam	4	17.7	63.6	0.28	0.11	Viets *et al.* 1954
		8.2		4	16.9	67.6	0.25	0.097	
	Tuber	7	N.R.	4	25	24.7	1	0.3	Dudka *et al.* 1996
		7.2		4	40	465	0.086	0.036	
		6.9		4	77	1,030	0.075	0.031	
		7.4		4	121	5,900	0.021	0.009	
		7.4		4	172	11,375	0.015	0.0065	
WOODY PLANTS									
Red Maple seedlings *(Acer rubrum)*	Whole plant	3.75	sandy peat	5	137	37.1	3.69	0.67	Vedagiri and Ehrenfeld 1991
		5.25		5	280	328.9	0.85	0.27	
		4.75		5	225	212.8	1.06	0.31	
		3.75		7	37.2	37.1	1	0.3	
		5.25		7	55.8	328.9	0.17	0.068	
		4.75		7	228.8	212.8	1.08	0.32	
Cranberry *(Vaccinium macrocarpon)*	Whole plant	3.75	sandy peat	5	180	37.1	4.85	0.77	Vedagiri and Ehrenfeld 1991
		5.25		5	52	328.9	0.16	0.064	
		4.75		5	60	212.8	0.28	0.11	
Alder	Leaves	4.3	sandy loam	6	53	80	0.66	0.22	Hogan and Wotton

Species	Tissue Type	pH	Soil Type	n	Zinc in Tissue (mg/kg dw)	Zinc in Soil (mg/kg dw)	BCF†	Log (BCF+1)‡	Reference
(Alnus sp.)		4.6		6	69	90	0.77	0.25	1984
		4.2		6	289	627	0.46	0.16	
		4.4		6	226	207	1.09	0.32	
		4.6		6	145	87	1.67	0.43	
		5.5		6	55	80	0.69	0.23	
		4.9		6	145	153	0.95	0.29	
Labrador Tea *(Ledum groenlandicum)*	Leaves	4.2	sandy	6	324	627	0.52	0.18	Hogan and Wotton 1984
		4.4		6	166	207	0.8	0.26	
		4.6		6	223	87	2.56	0.55	
		5.5		6	60	80	0.75	0.24	
		6.9	organic	6	376	2,133	0.18	0.072	
		4.9		6	150	153	0.98	0.3	
		4.3		6	74	80	0.93	0.29	
		4.6		6	61	90	0.68	0.23	
Jack Pine (*Pinus banksiana)*	Leaves	4.2	sandy loam	6	363	627	0.58	0.2	Hogan and Wotton 1984
		4.4		6	294	207	1.42	0.38	
		4.6		6	201	87	2.31	0.52	
		5.5		6	95	80	1.19	0.34	
		4.9		6	184	153	1.2	0.34	
		4.3		6	137	80	1.71	0.43	
		4.6		6	75	90	0.83	0.26	
Black Spruce (*Picea mariana*)	Leaves	4.2	sandy loam	6	227	627	0.36	0.13	Hogan and Wotton 1984
		4.4		6	165	207	0.8	0.26	
		4.6		6	163	87	1.87	0.46	
		5.5		6	76	80	0.95	0.29	
		4.9		6	132	153	0.86	0.27	
		4.3		6	120	80	1.5	0.4	
		4.6		6	62	90	0.69	0.23	
European aspen (*Populus tremula*)	Leaves	6.7	loamy sand	3	486	8,655	0.056	0.024	Krpata *et al.* 2008
		7.1		2	957.3	8,177	0.12	0.049	
		7.28		3	974	11,239	0.087	0.036	
		6.55		4	340	571	0.6	0.2	

† BCF = bioconcentration factor from soil to plants (see part B, Section 7.6.1.6, CCME 2006)
‡ log BCF = log (bioconcentration factor + 1)
NR = not reported

Appendix XVII. Summary of Statistical Measures for the BCFs Surveyed for Zinc

	n	Mean	Stan. Dev	Minimum	25%ile	Median	75%ile	Maximum	Range
leaves	58	0.65		0.0095	0.225	0.496	0.853	2.56	2.5505
transformed	58	0.20	0.132941407	0.0041	0.088	*0.17*	*0.268*	*0.55*	0.5459
shoots	51	0.529384314		0.0087	0.12	0.27	0.57	3.3	3.2913
transformed	*51*	*0.15314902*	0.15193825	*0.0038*	*0.049*	*0.1*	*0.195*	*0.63*	0.6262
roots	28	0.707785714		0.015	0.2425	0.505	1.025	2.8	2.785
transformed	*28*	*0.206339286*	0.15005333	*0.0065*	*0.094*	*0.18*	*0.305*	*0.58*	0.5735
grains	5	0.27176		0.0051	0.0097	0.052	0.092	1.2	1.1949
transformed	5	0.08128	0.14536015	0.0022	0.0042	0.022	0.038	0.34	0.3378
whole plant	9	1.46		0.16	0.28	1	0.28	4.85	4.69
transformed	*9*	*0.320222222*	0.25028473	*0.064*	*0.11*	*0.3*	*0.11*	*0.77*	0.706
Total	**151**								

The geometric mean of all BCFs is 0.26, calculated by using the log BCF and then back-transforming.

Appendix XVIII. Alternative Approach for Calculating Human Health Soil Quality Guidelines for Zinc when EDI>TDI.

Overview

For the purposes of soil quality guidelines (SQG) derivation, the recommended procedure when the estimated daily intake is larger than the tolerable daily intake (EDI>TDI) is to:

1. *Ensure the mean or best estimate of the mean EDI and TDI chosen are appropriate.*
2. *Calculate the SQG based on the 10 per cent EDI equation.*
3. *Calculate the SQG based on the 20 per cent TDI equation.*
4. *Choose the lower of the 10 per cent EDI or 20 per cent TDI calculated value and compare it to the background soil concentration (BSC).*
5. *If the lowest of the calculated value chosen from the 10 per cent EDI or 20 per cent TDI is greater than the BSC, use this value as the provisional SQG_{DH}. If the calculated value is less than the BSC, set the SQG_{DH} to the BSC. (See attached figure for pictorial explanation. The blue path shows the approach recommended in this document, the yellow path illustrates the CCME protocol when TDI > EDI, and the green path provides additional recommendations when the EDI is >90 per cent of the TDI.)*

Explanation for the removal of the soil allocation factor

The EDI[2] terms consider the contribution of background soils and the soil allocation factor as part of their calculation. Removing the soil allocation factor (SAF) is similar to setting the SAF to 1 instead of 0.2. Equation 2 incorporates the background soil and soil allocation factor in the 10 per cent EDI term. If a soil allocation factor of <1 is used, this would decrease the allowable EDI contribution from soils to 0.1 x SAF. For example, if the default SAF of 0.2 is used, this

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[2] The EDI is based on the sum of estimated human exposure to a substance through contact with various media (food, drinking water, soil, air, consumer products and dust). For illustrative purposes, the soil dermal EDI can be calculated using the following equation:

Where:

$$ug/kg/d = \frac{\left[\left(C_s \times SA_H \times SL_H\right) + \left(C_s \times SA_A \times SL_A\right) + \left(C_S \times SA_L \times SL_L\right)\right] \times RAF_{derm} \times EF \times 1000 ug/mg}{BW}$$

C_s = concentration of substance in soil (mg/kg)
SA = surface area for hands, arms and legs (cm^2)
SL = soil loading for hands, arms and legs (kg/cm^2/event)
RAF_{derm} = Relative Dermal Absorption Factor (unitless)
EF = event frequency (1 event/d)
BW = body weight (kg)

results in an EDI contribution from soil of 0.02 of the total EDI. The use of an SAF of 1 results in an EDI contribution from soil of up to 0.10 of the total EDI. It seems reasonable that a 10 per cent increase in EDI is within the variability of observed data from various media. (The GSC reported a provisional range of range of <2 to 1,770 mg/kg Zn in background till concentrations [Rencz et al. 2006; Grunsky 2010a]).

Justification for the use of equation #2.

Equations 1, 2 and 3 (described below) all follow the general CCME SQG equation with the (TDI–EDI) term replaced with a 0.1 EDI term. Equation 2 is the simplest of the 10 per cent EDI equations. The BSC can be removed from the equation because the EDIs are already above the TDI in cases where this approach is considered, and the addition of the BSC will not add any more precision to the calculation. If the value derived using Equation 2 is above the BSC, the SQG is set to the BSC by default.

Comparing SQGs derived for Zn using the three equations, the resulting SQGs from equations 1, 2 and 3 are similar, with the differences mainly attributed to the exclusion of the BSC from the equation.

Background

CCME (2006) outlines a protocol that should be used in the derivation of environmental and human health soil quality guidelines (SQGs). For threshold substances, the CCME equation used to derive the human health soil quality guideline takes the general form:

$$SQG_{DH} = \frac{(TDI - EDI) \times SAF \times BW}{IR \times AF \times ET} + BSC$$

where:
SQG_{DH} = human health soil quality guideline
TDI = tolerable daily intake (µg/kg bw/day)
EDI = estimated daily intake (µg/kg bw/day)
SAF = soil allocation factor (unitless)
BW = body weight (kg)
IR = ingestion rate of medium of concern (kg/day)
AF = absorption factor for medium of concern (unitless)
ET = exposure Term (unitless)
BSC = background soil concentration
RTDI = (TDI–EDI) residual tolerable daily intake (µg/kg bw/day)

To derive the guidelines for threshold substances, it is necessary to assign an allowable proportion of the total chemical exposure to the soil medium in the equation listed above. The TDI represents the total dose to which it is believed a human receptor can be safely exposed continuously over a lifetime without any deleterious effects. The EDI is an estimate of the total background exposure of human receptors to the substance, based on a multi-media exposure assessment.

CCME considers five primary media (i.e., air, water, soil, food and consumer products) to which people are potentially exposed. CCME proposed that a default value of 20 per cent (0.2) be

allotted to each of the five exposure media. For the purposes of deriving soil quality guidelines, 20 per cent the RTDI is apportioned to soils so that the SAF is arbitrarily set at 0.2 and allows for 80 per cent of the total incremental exposure from other media (i.e., food, air, water and consumer products). However, some soil contaminants may not be normally present in one or more of the other exposure media. If it can be proven that exposure to one or more of the remaining media are not relevant for the substance, the SAF may be adjusted upward from the 0.2 default by dividing the total exposure (100 per cent) by the number of applicable exposure media so that:

$$SAF = \frac{100\%}{\#\ of\ applicable\ media}$$

For some substances where the EDI exceeds the TDI, the equation used by CCME to derive the human health soil quality guideline would not apply, as the SQG derived would result in a negative number. In these cases, the CCME (2006) protocol states that:

> When the EDI is greater than the TDI (RTDI = 0), theoretically the population cannot be safely subjected to any increased exposure. In these cases, the provisional soil quality guideline should be set at the background soil concentration or practical quantification limit for that contaminant.

Issue

During the course of developing SQGs, some substances were found to have estimated background exposure rates greater than the toxicity benchmarks established for human health. If the EDI exceeds TDI, this implies that exposure to typical background levels of the substance exceeds a dose considered protective of human health. However, this does not necessarily mean that health effects are expected in the population at large because there is usually considerable uncertainty in the EDI and TDI values. EDIs are derived from estimates of the mean concentrations of the substance in exposure media (i.e. air, water, soil and food) and estimated mean or typical intake rates of the substance of concern via the various exposure media. Uncertainty in the mean EDI or best estimate of the mean EDI can arise from various sources such as limited availability of data for chemical concentrations in various media and the lack of or uncertainty in intake rates for food, water, air, and soil. In the case of food intakes, assumptions employed by the CCME may not reflect more current consumption patterns (Meridian Environmental Inc. 2007).

In situations where EDI>TDI, human health SQGs have either been set at background soil concentration levels or they have not been established at all. CCME (2006) does recognize that this may result in a fairly restrictive criterion and, as a result, they suggest that any models used to develop the EDI should be checked to ensure their accuracy and to assess any regional or site-specific factors. In cases where EDI>TDI, establishing human health soil remediation guidelines to background soil concentrations or practical quantification limits may not be pragmatic or practical and may result in high remediation costs at sites without any significant benefit to the protection of human health.

Approach

Recognizing that setting a human health SQG to background soil concentration levels or practical quantification limits may not be practical, a draft supplemental guidance document that outlined a general approach that could be used in cases when EDI>TDI was prepared (Meridian Environmental Inc. 2007).

Independent to the work completed by Meridian Environmental Inc., Wilson Scientific Consulting Inc. (2009) also addressed this issue for two substances (nickel and zinc). The processes and equations derived independently by Meridian Environmental Inc. and Wilson Scientific Consulting Inc. were very similar. Based on this work, an alternative to the approach outlined in the CCME (2006) protocol document for establishing human health SQGs in cases where the EDI>TDI for threshold substances and where exposure to soil is a minor contributor to the EDI was proposed.

Soil quality guidelines for nickel (Ni) and zinc (Zn) are used as examples of SQGs that could be derived using the options presented below for critical receptors (typically a toddler for residential and commercial exposure scenarios). If the EDI<TDI for the relevant scenario and receptor group, then the standard CCME equation applies and is used to derive the human health SQG for that scenario. For Ni and Zn, the standard CCME (2006) SQG_{DH} equation would apply for industrial sites since the adult is considered the critical receptor and, in that scenario, the TDI for both of these substances is greater than the EDI for the adult receptor.

For the scenarios where the EDI>TDI, the "TDI−EDI" term, also known as the Residual TDI (RTDI), in the CCME human health SQG equation, was modified and replaced with 10 per cent EDI. EDI estimates are subject to uncertainty and variability of the data upon which the EDI estimate is based. This includes uncertainty due to limited data on chemical concentrations in some environmental media and intake rates. In addition to uncertainty, chemical concentrations in various media and intake rates can be highly variable. Background concentrations can vary by orders of magnitude between regions in some media. For example, Grunsky *et al.* (2010a) report a mean Zn soil concentration of 48.1 mg/kg, a median concentration of 34 mg/kg, and a provisional range of <2 to 1,770 mg/kg in background soil concentrations.

Most other jurisdictions apply a target hazard index of 0.2 to the TDI, which is assumed to be sufficiently protective, irrespective of background exposure and exposure through other media at the site, when setting soil quality guidelines for inorganic substances. The 20 per cent TDI equation was included to allow comparison with the SQG calculated using 10 per cent EDI.

Exposure through food ingestion, which can comprise the largest portion of the EDI, is affected by variability in chemical concentrations within and between food types as well as the variability in diet compositions between individuals. A cursory examination of the estimated total daily intake of zinc from food by age class from the 2000–2007 Canadian Total Diet Study shows that food contributes over 97 per cent of the total EDI and standard deviations of mean intakes range from approximately 13 to 59 per cent, with the highest variability in intake rates in infants and toddlers. Based on the variability seen in food data and soil data, a 10 per cent increase in the EDI appears to be well within the variability observed in the data from various media. Sigal *et al.* (2006) conducted a probabilistic evaluation of EDIs for three metals and they found that the 95th percentile EDI was more than 50 per cent greater than the mean EDI in all cases. For most

naturally occurring substances, the contribution of soil ingestion to the EDI is relatively small (i.e., <1 per cent) compared to the contribution of exposure from food and water based on EDIs published by Health Canada (Meridian Environmental Inc. 2006). Therefore, given the variability and uncertainty in the EDI, an incremental increase in exposure of 10 per cent of the EDI is not expected to represent a biologically significant increase in exposure so long as the EDI represents a "typical" exposure and is not a worst-case exposure (Meridian Environmental Inc. 2006).

Equation 1 – (Meridian Environmental Inc. 2007)

Using the existing CCME human health SQG equation specified in the protocol document, Meridian Environmental Inc. modified the equation by replacing the RTDI term (i.e., TDI – EDI) with 0.1 EDI so the SQG equation becomes:

$$SQG_{DH} = \frac{(0.1 \times EDI) \times SAF \times BW}{IR \times AF \times ET} + BSC$$

The premise of this equation is that, based on a multi-media exposure assessment, a 10 per cent incremental increase in the mean EDI due to exposure to soil concentrations in excess of BSC is not expected to result in a significant shift in the range and frequency of EDI estimates across the population as a whole. Nor is it expected to result in any deleterious effects to human health. The contribution of soil to total exposure is often small (e.g., <1 per cent of EDI) relative to other media, and a small increase in soil concentration should only result in a small (perhaps negligible) increase in the EDI, so long as the EDI represents a "typical" exposure and not a worst-case exposure. The default SAF used in the CCME process is typically set at 0.2. However, as stated in CCME (2006), the SAF can be adjusted upwards if there is rationale that shows exposure to one of the five media listed is insignificant. In the case of Zn, it is proposed that consumer products would not be a significant source of Zn exposure on contaminated sites, relative to uncontaminated sites and that if this is a reasonable assumption, exposure to media on a contaminated site can be allocated as 0.25 to drinking water, 0.25 to food and 0.5 to direct exposure via soil ingestion, dermal contact and air.

Equation 2 – Wilson Scientific equation

The equation derived by Wilson Scientific Consulting Inc. is similar to the Meridian Environmental Inc. equation above except the BSC concentration and SAF were not included in the equation, so that:

$$SQG_{DH} = \frac{(0.1 \times EDI) \times BW}{IR \times AF \times ET}$$

The EDI term considers the contribution of background soils as part of its calculation, and, as such, the BSC was not included in the equation. In other words, the SAF is set to 1 and is included in the (0.1 x EDI) term. In cases where the contribution of soil to total exposure is not significant compared to the contribution relative to other media, the BSC is not expected to contribute significantly to the calculation of the SQG so that the derivation equation can be simplified by excluding the BSC and SAF terms from the equation.

Equation 3 – Modified equation

After review of the above proposed approaches, a modification was made based on the concept of using 10 per cent of the EDI when EDI>TDI, and including the BSC term in the SQG_{DH} derivation equation. One option was to subtract the soil contribution (EDI_{soil}) from the total EDI term since the contribution from the soil exposure pathway is accounted for in the BSC term in the equation so that:

$$SQG_{DH} = \frac{[0.1 \times (EDI_{total} - EDI_{soil})] \times SAF \times BW}{IR \times AF \times ET} + BSC$$

Various SAFs (0.2, 0.5 and 1.0) were applied in the equation to look at the variability of SQGs derived for Zn, based on the contribution of EDI from soil ranging from 2 to 10 per cent.

The SAF of 0.2 is the CCME default specified in the protocol document (CCME 2006) under normal circumstances and is included for comparative purposes. Using an SAF of 0.2 will result in a 2 per cent EDI in the numerator of the equation (e.g., 0.1x EDI x 0.2 x BW = 0.02 EDI x BW)

The use of an SAF of 0.5 will result in a 5 per cent EDI in the numerator of the equation (e.g., 0.1 x EDI x 0.5 x BW = 0.05 EDI x BW)

An SAF of 1 will result in a 10 per cent EDI in the numerator of the equation (e.g., 0.1 x EDI x 1 x BW = 0.1 EDI x BW).

Equation 4 – 20 per cent TDI

To develop soil quality guidelines for the protection of human health, most other jurisdictions apply a target hazard index of 0.2, which is assumed to be sufficiently protective irrespective of background exposure and exposure through other media at the site. In this equation, the background soil concentration is not included. The 20 per cent TDI equation is:

$$SQG_{DH} = \frac{(0.2 \times TDI) \times BW}{AF \times IR \times ET}$$

In any case where a proportion of the EDI is used, Meridian Environmental Inc. and Wilson Scientific Consulting Inc. both recommend that the above calculation should also be completed and that the SQG be based on the lower of the estimates.

Calculations

For Zn, the EDI exceeds the TDI for residential and commercial exposure scenarios where the toddler is considered the critical receptor. Using the equations listed above, residential and commercial soil quality guidelines were calculated using SAFs of 0.2, 0.5 and 1.0 for comparative purposes where applicable. The resulting values are listed in the tables below.

Residential scenario **Critical receptor: Toddler**	**Zn** **RSQG**
Equation 1 using SAF = 0.2	2,100
Equation 1 using SAF = 0.5	5,200

Equation 1 using SAF = 1.0	10,000
Equation 2 no BSC included	**10,000**
Equation 3 EDI_{soil} removed; SAF = 0.2	2,100
Equation 3 EDI_{soil} removed; SAF = 0.5	5,200
Equation 3 EDI_{soil} removed; SAF = 1.0	10,000
Equation 4 20% TDI	16,000

Commercial scenario **Critical receptor: Toddler**	**Zn** **CSQG**
Equation 1 using SAF = 0.2	3,200
Equation 1 using SAF = 0.5	8,000
Equation 1 using SAF = 1.0	16,000
Equation 2 no BSC included	**16,000**
Equation 3 EDI_{soil} removed; SAF = 0.2	3,200
Equation 3 EDI_{soil} removed; SAF = 0.5	8,000
Equation 3 EDI_{soil} removed; SAF = 1.0	16,000
Equation 4 20% TDI	24,000

Bold indicates equations and calculated values recommended for consideration when EDI>TDI

Industrial scenario **Critical receptor: Adult**	**Zn** **ISQG**
CCME equation	270,000
20% TDI	420,000

Discussion

For the example above, subtracting EDI_{soil} from the overall EDI does not affect the SQG because the contribution to the EDI from soil is not significant when compared to the exposure contribution from other media (i.e., food). This is shown by the resulting SQGs calculated using Equations 1 and 3, which are the same.

For the residential and commercial scenarios using toddlers as the critical receptor, all SQGs derived using the various 10 per cent EDI equations (Equations 1, 2 and 3) were less than the calculated SQG using the 20 per cent TDI equation (Equation 4). Therefore, an SQG calculated using any of these 10 per cent EDI equations would be more conservative than what is currently being done in other jurisdictions that use a 20 per cent TDI equation to derive SQGs. If an SAF of 1 is used, the SAF term essentially drops out of Equations 1 and 3. If, as stated earlier, we accept that a 10 per cent increase in the EDI is within the variability observed in the various media and does not represent a biologically significant increase in exposure, the SAF term can be eliminated from the equations without affecting the calculated SQG value significantly.

In cases where the EDI is quite large in comparison to the TDI, exposure based on 10 per cent of the EDI may result in exposure that exceeds the TDI; therefore, it is suggested that, as a further check on the appropriateness of the 10 per cent EDI equations, 20 per cent of the TDI be calculated and that the lower of the two values (either calculation based on 10 per cent of the

EDI or 20 per cent of the TDI) be used to establish the SQG. This approach is more conservative than any approach taken in other jurisdictions that establish SQGs for human health.

Recommended approach to setting Human Health Soil Quality Guidelines when mean EDI>TDI

The CCME equation that utilizes the TDI–EDI term is unique. Other jurisdictions that establish SQGs for human health typically derive SQGs based on 20 per cent of the TDI without considering exposure from background levels. In situations where the EDI>TDI, and the background soil concentration is not expected to contribute significantly to exposure, the 10 per cent EDI equation (Equation 2) should be considered.

For the purposes of soil quality guidelines derivation, the recommended procedure when EDI>TDI is to:

1. Ensure the mean or best estimate of the chosen mean EDI and TDI are appropriate.
2. Using Equation 2, calculate the SQG based on 10 per cent EDI.
3. Using Equation 4, calculate the SQG based on 20 per cent TDI.
4. Choose the lower of the 10 per cent EDI or 20 per cent TDI calculated value (i.e., the lower of the two values calculated from Equation 2 and 4) and compare it to the BSC.
5. If the lowest of the calculated value (from Equations 2 or 4) is greater than the BSC, use this value as the provisional SQG_{DH}. If the calculated value is less than the BSC, set the SQG_{DH} to the BSC.

Note that the discussion and recommendations outlined thus far only apply to direct contact pathways for inorganic threshold substances. In the case of Zn, these equations can be considered for oral, inhalation and dermal exposure pathways.

Other considerations:

During EDI>TDI discussions, another issue was brought up: What about when the EDI approaches the TDI but does not exceed the TDI? Specifically, when the EDI is within 90 per cent of the TDI, the term “TDI–EDI” can become quite small and result in a much lower SQG than if the EDI exceeded the TDI.

Consequently, when EDI is estimated to be greater than 90 per cent of the TDI, the following steps are recommended:

1. Calculate the SQG using the CCME (i.e., TDI–EDI) equation.
2. Calculate the SQG using 10 per cent EDI (Equation 2).
3. Calculate the SQG based on the 20 per cent TDI equation.
4. Compare the SQGs derived from the CCME equation (step 1) and the 10 per cent EDI equation (step 2) and choose the higher of the two.
5. Compare the SQG (step 4) with the SQG derived using the 20 per cent TDI equation (step 3) and use the lower of the two as the SQG.

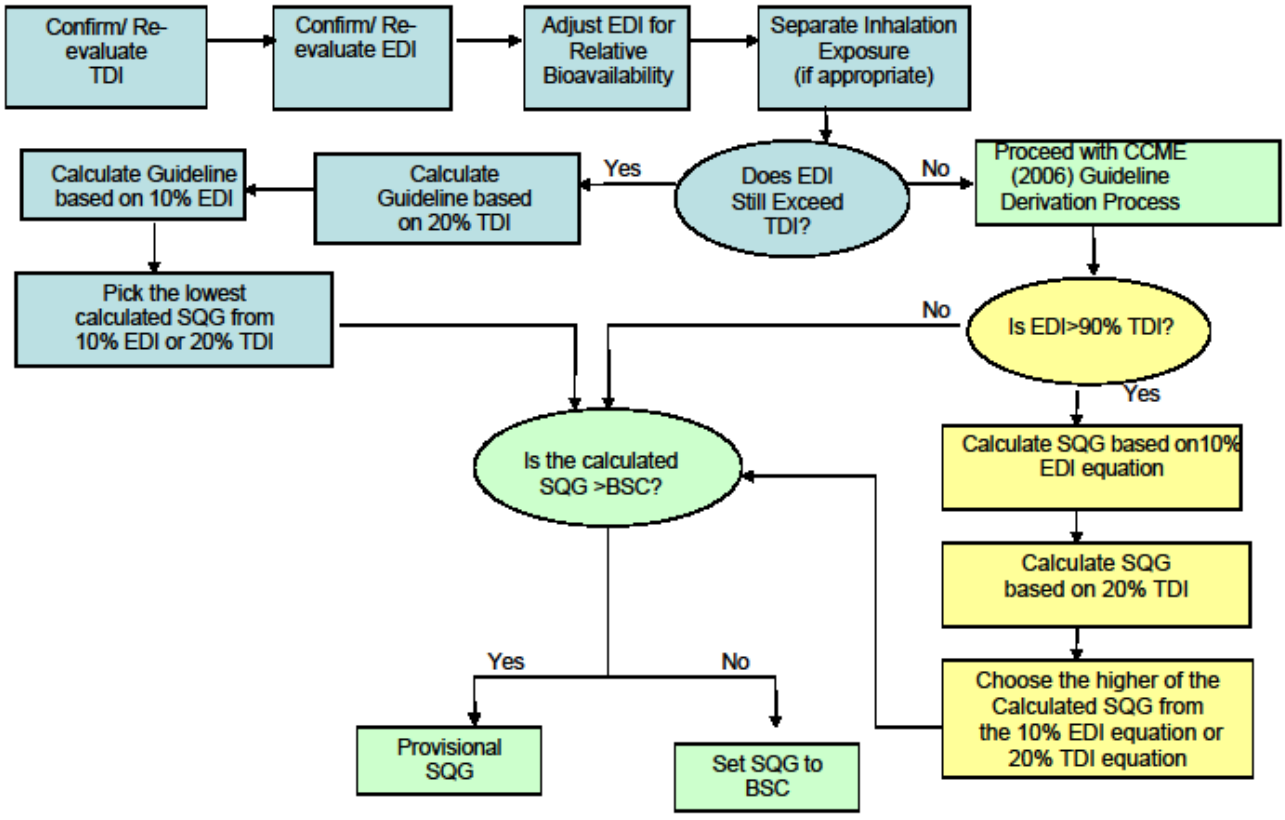
Due to the mathematics of the equations, it is possible to calculate a lower SQG using the "TDI–EDI" equation than if the SQG is calculated using one of the 10 per cent EDI equations in cases where the EDI is slightly less than the TDI. However, it is difficult to justify treating a substance less stringently because the EDI is slightly lower than the TDI. Intuitively, the SQG should increase as the EDI decreases. The last step of comparing the higher of the two calculated SQGs using the CCME equation and the 10 per cent EDI equation to the SQG calculated using the 20 per cent TDI equation ensures that the resulting SQG chosen is less than 20 per cent of the TDI, which is used by most other jurisdictions that derive SQGs.

The recommended procedure to follow when EDI is greater than TDI is summarized in Figure 1.

Other issues for future discussion:

- The equations as described do not apply to volatile, organic or non-threshold substances, or substances with multiple relevant exposure pathways (i.e., direct contact, vapour inhalation, drinking water). The equations would have to be modified and appropriate allocation factors would be determined and included in the equation to address relevant exposure pathways for those substances.

Figure 1: Approach for Deriving Human Health SQGs when EDI > TDI (for direct contact, inorganic threshold substances)



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