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Canadian Soil Quality Guidelines for

Inorganic Mercury: Environmental

Supporting Document — Final Draft
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NOTICE

This final draft document provides the information supporting the derivation of environmental soil quality guidelines for inorganic mercury. Development of these soil quality guidelines was initiated through the National Contaminated Sites Remediation Program (NCSRP) which officially ended in March 1995. Given the need for national soil quality guidelines for contaminated sites management and many other applications, development was pursued under the direction of the CCME Soil Quality Guidelines Task Group after the end of the NCRSP.

This document is a working document that was released shortly after the publication of "A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines" (CCME 1996). The CCME recognizes that some refinements or changes to the Protocol may become necessary upon application and testing. If required, amendments to the Protocol will be made and the guidelines will be modified accordingly. For this reason guidelines are referred to in this document as CCME Recommended Guidelines. Readers who wish to comment or provide suggestions on the Protocol or on the guidelines presented in this document should send them to the following address:

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This document is a supporting technical document. It is available in English only. A French Abstract is given on page vii.

Ce document technique de soutien n'est disponible qu'en anglais avec un résumé en français présenté à la page vii.

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ABSTRACT

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

This report was prepared by the Guidelines Division of the Science Policy and Environmental Quality Branch (Environment Canada), which acts as Technical Secretariat for the CCME Soil Quality Guidelines Task Group. The Guidelines were derived according to the procedures described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996).

Following the introduction, chapter 2 presents chemical and physical properties of mercury and a review of the sources and emissions in Canada. Chapter 3 discusses mercury's distribution and behavior in the environment while chapter 4 reports the toxicological effects of mercury on microbial processes, plants, and animals. These informations are used in chapter 5 to derive soil quality guidelines for mercury to protect environmental receptors in four types of land uses: agricultural, residential/parkland, commercial, and industrial.

The following soil quality guidelines are recommended by the CCME based on the available scientific data. For mercury, the environmental soil quality guideline (SQG_E) relative to agricultural and residential/parkland land uses is 10 $mg \cdot kg^{-1}$ soil, and it is 30 $mg \cdot kg^{-1}$ soil for commercial and industrial land uses. These environmental soil quality guidelines are optimized for soils within the pH range of 4 to 8.3 as the toxicological studies on which they are based were conducted within this pH range.

RÉSUMÉ

Les recommandations canadiennes pour la qualité de l'environnement, élaborées sous les auspices du Conseil Canadien des Ministres de l'Environnement (CCME), sont des concentrations ou des énoncés décrivant les limites recommandées dans le but d'assurer le maintien et le développement durable d'utilisations désignées des ressources. Les recommandations canadiennes pour la qualité des sols proposées par le CCME peuvent être utilisées comme base pour l'uniformisation des processus d'évaluation et d'assainissement des terrains contaminés au Canada.

Le présent document a été préparé par la Division des Recommandations de la Direction de la Qualité de l'Environnement et de la Politique Scientifique (Environnement Canada), qui agit comme secrétaire technique pour le Groupe de Travail du CCME sur les Recommandation pour la Qualité des Sols. Les 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 1996).

Faisant suite à une brève introduction, le chapitre 2 présente les propriétés physiques et chimiques du mercure de même qu'un survol des sources et des émissions au Canada. Le chapitre 3 discute du devenir et du comportement de cette substance dans l'environnement alors que le chapitre 4 rapporte ses effets toxicologique sur les processus microbiens, les plantes et les animaux. Ces informations sont utilisées au chapitre 5 afin d'élaborer des recommandations pour la qualité des sols relatives au mercure en vue de la protection de l'environnement dans le cadre de quatre types d'utilisations de terrains: agricole, résidentiel/parc, commercial et industriel.

Les recommandation pour la qualité des sols suivantes, proposées par le CCME, sont fondées sur les données scientifiques disponibles. Pour le mercure, la recommandation pour la qualité des sols en vue de la protection de l'environnement (RQS_E) relative aux terrains à vocation agricole et résidentielle/parc est de 10 $mg \cdot kg^{-1}$ de sol. Elle est de 30 $mg \cdot kg^{-1}$ de sol pour les terrains à vocation commerciale et industrielle. Ces recommandations pour la qualité des sols en vue de la protection de l'environnement sont à leur optimum dans des sols avec pH entre 4 et 8.3 puisque les études toxicologiques utilisées pour leur élaboration ont été effectuées dans ces mêmes conditions de pH.

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1. INTRODUCTION

The Canadian Council of Ministers of the Environment's (CCME) Canadian Environmental Quality Guidelines are numerical limits for contaminants intended to maintain, improve, or protect environmental quality and human health. CCME Canadian Soil Quality Guidelines can be used as the basis for consistent assessment and remediation of contaminants at sites in Canada along with CCME guidelines issued for the protection of water quality, sediment quality and tissue quality. In response to the urgent need to begin remediation of high priority "orphan" contaminated sites, an interim set of criteria was adopted from values currently in use in various jurisdictions across Canada (CCME 1991). Many of the CCME interim soil remediation criteria do not have a complete supporting scientific rationale and are being updated based on current scientific information.

This report reviews the sources and emissions of mercury, its distribution and behaviour in the environment, and its toxicological effects on terrestrial mammals, plants and soil organisms. This information is used to derive guidelines for inorganic mercury to protect ecological receptors according to the processes outlined in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996) for agricultural, residential/parkland, commercial and industrial land uses.

The values derived herein are environmental soil quality guidelines, and are intended as general guidance. Site specific conditions should be considered in the application of these values. The values may be applied differently in various jurisdictions, therefore, the reader should consult the appropriate jurisdiction for application of the values.

2. BACKGROUND INFORMATION

2.1 Physical and Chemical Properties

Mercury (quicksilver, *hydrargyrum*, liquid silver) is a dense silver-white metal which is liquid at room temperature and is characterized by low electrical resistivity, high surface tension, and high thermal conductivity (Andren and Nriagu 1979; Environment Canada 1981). The important physical and chemical properties of mercury compounds are presented in Table 1.

The two properties which largely determine the environmental behaviour of mercury are the high vapour pressure of liquid mercury, yielding hazardous vapour concentrations, and the solubility of organic forms. Mercury can exist in three stable oxidation states: elemental mercury (Hg^0/ Hg(0)), mercurous ion (Hg_2^{2+}/Hg(I)), and mercuric ion (Hg^{2+} /Hg(II)). Mercury can be oxidized to both inorganic and organic salts, such as chlorides and sulphates, and to organomercury compounds. A wide range of organomercury compounds are present in the environment and are characterized by the attachment of mercury to one or two carbon atoms to form compounds of the type R-Hg-X and R-Hg-R', where R and R' represent organic moieties, and X represents a halogen. The organic moieties may take the form of alkyl, phenyl and methoxyethyl radicals (WHO 1976). A subclass of short-chained alkylmercurials, which include monomethyl- (CH_3Hg^+) and dimethylmercury

($(CH_3)_2Hg$), are the predominant organic mercury compounds found in natural systems. Dimethylmercury is less stable and more volatile than monomethyl compounds (Environment Canada 1981). Other organic forms of mercury include Phenylmercuric acetate (PMA), Phenylmercuric chloride (PMC), Methylmercuric dicyanediamide (MMD), Methylmercuric acetate (MMA), and Methylmercuric chloride (MMC).

The vapour pressure of mercury is highly dependent on temperature. The tendency of liquid mercury to form small droplets increases its rate of evaporation. Mercury is found in the environment, not as the liquid metal, but mainly in the form of amalgams and inorganic salts which have lower vapour pressures than elemental mercury (Andren and Nriagu 1979). The solubility of mercury compounds in water increases in the order: elemental mercury < mercurous chloride < methylmercury chloride < mercuric chloride. The toxicity of the inorganic forms is generally less than that of the organic compounds. Within the inorganic forms, toxicity increases as lipid solubility increases (Halbach 1990).

2.2 Analytical Methods

There are a number of analytical methods available for measuring mercury in soils. The CCME (1993) recommended method is the Manual Cold Vapor Technique of US EPA (1983) Method 7471A, Rev.1 (SW-846). This method is applicable to the determination of total mercury (organic and inorganic) in soils, sediments and solid wastes. All samples are dissolved prior to analysis and analyzed using an atomic absorption spectrometer equipped with a cold vapor system and a mercury hollow cathode lamp. The method is based on the absorption of radiation at 253.8 nm by mercury vapor. The detection limit for this method is 0.0002 $mg \cdot L^{-1}$. Following corrections for a 1g soil sample digested, with the final extract being diluted to 100 ml, a detection limit of 0.02 mg $Hg \cdot kg^{-1}$ soil would be calculated.

Other common methods for total mercury include hot digestion of soil samples with HCl, HNO_3 and $KMnO_4$, and analyzing the extracts with inductively coupled plasma (ICP) spectroscopy (Sheppard *et al.* 1993) or with atomic absorption spectrophotometry. Additional methods are being developed by the National Laboratory for Environmental Testing, Burlington, Ontario.

2.3 Production and Uses in Canada

A number of mercury compounds are widely used in agriculture, medicine, and industry in Canada and have contributed to environmental contamination. Table 2 presents a brief summary of mercury compounds and their uses. Mercury emissions by sector are presented in Table 3. Total Canadian anthropogenic emissions to the atmosphere were estimated at 39,855 kg with the major source being base metal recovery (40%) (Environment Canada 1981).

2.4 Levels in the Canadian Environment

Mercury is a naturally occurring element often found in association with sulphur in mineral ores and rocks. Mercury has been extensively mined and used for over 2000 years. Consequently, there is widespread environmental contamination with mercury. Mercury in terrestrial environments is redistributed to the atmosphere through vaporization, volcanic action and weathering of the earth's crust (Environment Canada 1983).

Mercury is found in almost every environmental matrix including air, volcanic gases, fresh water, sea water, soils, mineral ores, lake and river sediments, and living organisms including plants, animals, and humans.

Soil

Concentrations of mercury found in Canadian soils are summarized in Table 4.

The general average terrestrial concentration of mercury in the Canadian environment is in the range of 0.01 to 0.4 $mg \cdot kg^{-1}$ (Environment Canada 1979; Jonasson and Boyle 1972; McKeague and Kloosterman 1974; Gracey and Stewart 1974; OMEE 1994) except in areas of ore deposits, spills, landfills, and accidents at metal processing plants. Frank *et al.* 1976 observed mercury concentrations ranging from 0.01 to 1.14 $mg \cdot kg^{-1}$ in Ontario agricultural soils, with a mean of 0.11 mg $Hg \cdot kg^{-1}$. Recently background levels for Ontario have been established at 0.16 $mg \cdot kg^{-1}$ for agricultural sites and 0.23 $mg \cdot kg^{-1}$ for residential/ parkland sites (OMEE 1994). In Alberta soils, background mercury levels ranging from 0.01 to 0.135 $mg \cdot kg^{-1}$ have been reported (Dudas and Cannon 1983; Dudas and Pawluk 1976; George *et al.* 1994). Elevated mercury levels are common in British Columbia due to the cinnabar deposits. High soil mercury levels are also reported in Quebec and Ontario near areas of known gold, copper or zinc mineralization (Environment Canada 1979). Jonasson and Boyle (1972) reported high mercury concentrations in some raw fossil fuels. Numbers range from 0.01 mg $Hg \cdot kg^{-1}$ for some coals, to 900 mg $Hg \cdot kg^{-1}$ for some bitumens, asphalts and solid hydrocarbons. According to the authors, these numbers reflect the strong binding exerted by organic chelates on mercury and the ability of organic material to accumulate heavy metals.

Sediments

Sediments are often the repository for mercury compounds in aquatic environments. Mercury can be rendered virtually inactive in deep sediments or, in surface sediments, it can be slowly converted to methylmercury, principally by sulphur-reducing bacteria, desorbed, rapidly picked up by living organisms and bioaccumulated (Bigham and Henry 1993; ENVIRO TIPS 1984).

In Ontario, mercury levels in sediments ranged from 0.08 to 0.93 $mg \cdot kg^{-1}$ for Big Creek Marsh (Mudroch 1980), from 0 to 0.3 $mg \cdot kg^{-1}$ for Tadenac lake (Wren and MacCrimmon 1986), from 0.15 to 0.83 $mg \cdot kg^{-1}$ for the Great Lakes (Thomas and Jaquet 1976) but from 0.29 to 1.62 $mg \cdot kg^{-1}$ for lake Erie (Thomas and Jaquet 1976). A value of 0.03 mg $Hg \cdot kg^{-1}$ was reported for lake Wabigoon (Rudd and Turner 1983). In Manitoba, mercury levels in the sediments of northen lakes range from

0.03 to 0.21 $mg \cdot kg^{-1}$ (Jackson 1988) whereas lakes from the Flin Flon area have reported mercury concentrations in sediments of 3.27, 3.77, 6.39 and 9.22 $mg \cdot kg^{-1}$ for Hamel, Douglas, Cliff and Phantom lakes respectively (Harrison and Klaverkamp 1990). In Saskatchewan, levels from 0.05 to 0.09 mg $Hg \cdot kg^{-1}$ dw sediments have been reported for Buffalo Pound lake (Hammer *et al.* 1988).

La Have river in Nova Scotia had mercury levels in sediments ranging from 0.32 to 3.54 $mg \cdot kg^{-1}$ (Cranston 1976). In Alberta, the South Saskatchewan, Bow, Red Deer and Oldman rivers had levels of 0.04, 0.04, 0.05 and 0.06 mg $Hg \cdot kg^{-1}$ sediments, respectively (George *et al.* 1994). Also, the North Saskatchewan river was reported to have sediment mercury levels below or equal to 0.1 $mg \cdot kg^{-1}$ (Ramamoorthy *et al.* 1985).

In British Columbia, sediment mercury levels of 0.095, 0.08, 0.2 and 0.13 $mg \cdot kg^{-1}$ have been reported for Barkley Sound, Quatsino Sound, Laredo Sound and Surf Inlet, respectively (Harding and Goyette 1989).

Water

There are relatively few data available on mercury concentrations in surface waters. Many earlier reports gave levels in water as low as the detection limits which, generally, ranged from 10 to 50 $ng \cdot L^{-1}$ (NRCC 1979). Median concentrations of mercury in Canadian drinking water from three provinces have been reported to be < 290 $ng \cdot L^{-1}$ (Health and Welfare Canada 1980). Point source contamination of surface waters with mercury, in Canada, has largely been the result of the use of mercury as a fungicide in the pulp and paper and chlor-alkali industries.

In an acidified watershed in central Ontario, Mierle (1990) observed a positive correlation between aqueous mercury and dissolved organic carbon. Total mercury concentrations were generally <5 $ng \cdot L^{-1}$ but during low flow periods, they exceeded 20 $ng \cdot L^{-1}$. Several recent studies indicate that background mercury concentrations range from 1 to 20 $ng \cdot L^{-1}$ in freshwater (Bloom 1989; Kudo *et al.* 1982; Mierle 1990).

The Wabigoon River in northwestern Ontario received an estimated 10 metric tons of mercury from a chlor-alkali operation in Dryden (Jackson *et al.* 1982). Total mercury levels in water up to 370 $ng \cdot L^{-1}$ were reported, with levels in the receiving waters typically in the 20-40 $ng \cdot L^{-1}$ range (Parks *et al.* 1989). A mean mercury level of 190 $ng \cdot L^{-1}$ in the mill effluent was reported by COSC (1983).

In the North Saskatchewan river in Alberta, Ramamoorthy *et al* (1985) reported 90 $ng \cdot L^{-1}$ in upstream Edmonton and a mean of 200 $ng \cdot L^{-1}$ in the downstream.

Air

Elemental mercury and different forms of divalent mercury in gaseous or particulate phases are the most frequently emitted forms to air by anthropogenic emissions (OECD 1993).

There is little information available on Canadian mercury levels in air. Generally, air concentrations have been measured in areas where point sources exist. Ambient air mercury levels are typically low and range from less than 1 $ng \cdot m^{-3}$ in rural areas to several thousand $ng^{-3}m$ in urban areas (Environment Canada 1979). Wiebe (1978) reported levels of gaseous mercury below 2 $ng \cdot m^{-3}$, in samples collected by airplane over Southern Ontario. At point sources, concentrations of mercury in Canadian air have been reported to be as high as 64.3 $\mu g \cdot m^{-3}$ (Environment Canada 1979).

Barton *et al.* (1980) reported ambient air concentrations of 1 to 10 $ng \cdot m^{-3}$, elemental mercury being the main constituent. Particulate mercury accounted for less than 10% of the total mercury concentration. Organo-mercury vapour did not contribute significantly to total mercury.

2.5 Existing Criteria and Guidelines

Existing criteria and guidelines for mercury in soil have been established for a number of jurisdictions and are presented in Table 5.

3. ENVIRONMENTAL FATE AND BEHAVIOUR

The major soil factors that determine the fate and behaviour of mercury are pH, organic matter and clay content, CEC, aeration and texture. The major processes that determine the mobility and distribution of mercury in the terrestrial environment are adsorption, chemical reactions, leaching, volatilization, photolysis and biodegradation. These processes are dependant on the soil factors mentionned above. The following paragraphs summarize the relationship between processes and factors:

Adsorption

Adsorption is the dominant process determining mercury's fate in the terrestrial environment (Hogg et al. 1978). It depends on mercury's chemical form, soil pH, colloids, CEC and redox potential (Hogg et al. 1978, Kabata-Pendias and Pendias 1992). Adsorption is increased by the presence of organic matter due to mercury complexation with humic and fulvic acids, therefore it is greater in surface horizons due to high humus content (Lodenius et al. 1987). Adsorption is maximized at pH 4-5 (Semu et al. 1986, Thanabalasingam and Pickering 1985), is decreased by Cl^- ions (Andersson 1979). In neutral and low organic matter soils, it depends on iron and clay minerals as important adsorption sites (OECD 1993). When added in elemental, cationic or anionic forms, mercury strongly adsorbs to soils (Kabata-Pendias and Pendias 1992).

Chemical Reactions

Mercury undergoes methylation by aerobic and anaerobic bacteria (NRCC 1979, WHO 1991). Methyl mercury species are mobile, bioavailable and highly toxic (Lexmond et al. 1976, Bigham and Henry 1993). Under reduced conditions, mercury and sulphide ions form HgS, which is insoluble and

resistant to methylation (NRCC 1979). Under aerobic conditions, HgS is oxidized to the sulphate form $HgSO_4$, which can udergo methylation (NRCC 1979). Bacterial action can cause demethylation of methyl mercury compounds (NRCC 1979). Chloride concentration and pH determine the chemical form of monomethylmercuric ion complexes (NRCC 1979).

Leaching

Leaching of mercury occurs in soils with little or no organic matter and light texture (Kabata-Pendias and Pendias 1992). Chloride complexes of mercury are soluble and hence, subject to leaching (Kabata-Pendias and Pendias 1992). In acid soils, mercury is leached out in a bound form with organic matter whereas in neutral and alkaline soils, mercury is leached out in an inorganic active form (Kabata-Pendias and Pendias 1992). Acid rain increases leaching of mercury (Lodenius et al. 1987). Highest leaching of mercury occurs in spring and autumn (Jonasson and Boyle 1972).

Volatilization

Volatilization is due to chemical and biological transformations of mercury compounds in soils (Frear and Dills 1967, Rogers and McFarlane 1979). The main form of mercury in the air is elemental mercury, but dimethylmercury may also occur (Lindberg et al. 1987, WHO 1989). Volatilization increases with pH and temperature but also depends on organic matter content, redox potential, moisture content and porespace of the soil (Frear and Dills 1967). Under favourable conditions, 50% of added mercury has volatilized within a week (Gilmour and Miller 1973).

Photolysis

Monomethylmercury may decompose photolytically to elemental mercury and methyl radicals (NRCC 1979). Dimethylmercury may be transformed to the mono form at low pH by ultraviolet light (Jernelov 1975). Photolysis of mercury chloride compounds may result in the production of methylmercury ions, dimethylmercury and metallic mercury (Jewett et al. 1975).

Biodegradation

Mercury is subject to biotic and abiotic transformations in soils (Kabata-Pendias and Pendias 1992, Andersson 1979). Both aerobic and anaerobic soil microbes can convert elemental mercury to mono and then to dimethylmercury (Kabata-Pendias and Pendias 1992). On the other hand, soil microbes are also capable of converting organic mercury to elemental mercury, but this is a slower process (Kabata-Pendias and Pendias 1992). *Clostridium* species, under anaerobic conditions, transform several mercury species (except HgS) to methylmercury (Yamada and Tonomura 1972). Pseudomonas species are mainly responsible for degrading organic and inorganic mercurials to metallic mercury (Furukawa and Tonomura 1972). Methylmercury is stable in soils (Rundgren et al. 1992).

4. BEHAVIOUR AND EFFECTS IN BIOTA

4.1 Soil Microbial Processes

The available information on the toxicity of mercury to terrestrial microbial processes is presented in Tables 6 and 7. Table 6 regroups the toxicity studies selected for use in soil quality guidelines derivation whereas table 7 presents other consulted studies. The selection process is described in section 5.1. The consulted and selected studies show that the toxicity of mercury, depending on the type of soil and microorganisms, varies from a few milligrams to thousands of mg $Hg \cdot kg^{-1}$ soil.

Von Stadelmann and Santschi-Fuhrimann (1987) reported that the lowest content of total mercury at which effects were detected (25% reduction in microbial respiration) is in the range of 0.06 to 0.08 mg $Hg \cdot kg^{-1}$ soil. Wilke (1988) observed similar effects at 1.3 mg $Hg \cdot kg^{-1}$ in soils. The soluble fractions in both studies were the same, 0.02 mg $Hg \cdot kg^{-1}$ (Lindqvist 1992).

Van Faassen (1973) reported a 20% reduction of CO_2 evolution in a clay soil and in a dune sand, when 74 mg $Hg \cdot kg^{-1}$ were added. Following the addition of 10 and 100 mg $Hg \cdot kg^{-1}$ (as mercury sulfate) to an acidic sandy soil, significant reductions in CO_2 release (29 to 55%) were observed after 2 and 8 weeks incubation periods (Cornfield 1977).

Landa and Fang (1978) observed that 200 mg $Hg \cdot kg^{-1}$ soil were required for a significant inhibition of carbon mineralization in short-term studies (6 hours). Whereas, for a chronic exposure period of 4 weeks, levels of only 0.1 to 10 $mg \cdot kg^{-1}$ soil, depending on soil type, were necessary for significant inhibition.

Zelles *et al.* (1985, 1986) reported NOECs at 1.47 to 14.78 mg $Hg \cdot kg^{-1}$ and a significant reduction in microbial respiration (CO_2 production) at 14.78 to 147.8 mg $Hg \cdot kg^{-1}$ in different clay and sand soils.

Tu (1988) reported the NOEC for microbial respiration at 70 mg $Hg \cdot kg^{-1}$, and a significant decrease at 140 mg $Hg \cdot kg^{-1}$ in a sandy loam. Spalding (1979) observed no effect on enzyme activity at 10 to 100 mg $Hg \cdot kg^{-1}$, but at 100 to 1000 mg $Hg \cdot kg^{-1}$ significant decreases were observed in litter incubated for 28 days.

Van Faassen (1973) reported, for nitrogen mineralization, a NOEC of 7.4 mg $Hg \cdot kg^{-1}$ when using $HgCl_2$ in sandy and in clay soils. With PMA the same author observed no effects in clay soils and a 57% inhibition in sandy soils at 6 mg $Hg \cdot kg^{-1}$. Liang and Tabatabai (1977, 1978) reported a significant inhibition of nitrification by $HgCl_2$ in different soils at 1003 mg $Hg \cdot kg^{-1}$. Wilke (1989) reported EC_{26} and EC_{85} at 50 and 200 mg $Hg \cdot kg^{-1}$, respectively, on nitrogen mineralization in soils.

Rogers and Pryfogle (1986) estimated $(P_2)_{50}$ values for enzyme activity in fine sandy loam. The $(P_2)_{50}$ is an index analogous to the LC_{50} but is a measurement of reduction in rate of reaction rather than lethality. The observed $(P_2)_{50}$ values were 26 $mg \cdot kg^{-1}$ for $Hg(NO_3)_2$ and 36 $mg \cdot kg^{-1}$ for $HgCl_2$.

Regarding urease activity in soils, Tabatabai (1977) reported 25 to 29% inhibition at 100 mg $Hg \cdot kg^{-1}$ soil and 75 to 98% inhibition at 1003 mg $Hg \cdot kg^{-1}$ soil added as $HgCl_2$. Similar results were obtained by Spalding (1979), who measured extractable enzyme activity in Douglas-Fir needle litter. Following the addition of $HgCl_2$, the NOEC values ranged from 10 to 100 mg $Hg \cdot kg^{-1}$ dw and enzyme activity was depressed at 100 to 1000 mg $Hg \cdot kg^{-1}$ dw for cellulase, xylanase, amylase, invertase, beta-glucosidase and polyphenoloxidase activity.

Bremner and Douglas (1971) reported that the inhibition of urease activity varied with the form of mercury and type of soil. In general, the inhibition varied from 36 to 81% depending on the soil, form of mercury ($HgCl_2$, $HgSO_4$ or PMA) and concentration (30, 60, 120 or 180 mg $Hg \cdot kg^{-1}$).

4.2 Terrestrial Plants

Uptake Metabolism and Elimination

Mercury and its compounds are absorbed by plant roots and, to a limited extent, translocated to plant organs. Generally, the accumulation of mercury in plants increases with increasing soil mercury concentration. Significant increases in concentration of mercury in plants were observed in mercury treated soils compared to untreated soils (Gracey and Stewart 1974; Lenka *et al.* 1992). However, soil type and soil organic matter content influence this process due to the strong adsorption of mercury compounds in soils.

Mercury is also absorbed by plants through the stomata from the atmosphere (Hg^0 and, partially, ionic mercury). This is particularly influenced by illumination (Browne and Fang 1978).

Bioaccumulation

Plants accumulate mercury in roots, foliage, stems, pods and leaves (Gracey and Stewart 1974; Hogg *et al.* 1978; John 1972). The form of mercury present in soil influences absorption and transport in plants. Conflicting observations were reported regarding the forms absorbed by plants. However, it appears that MMC (CH_3HgCl) is more available than other forms such as $HgCl$, PMA, and mercuric sulphate (Godbold 1991; Hogg *et al.* 1978). Some authors reported no significant differences in plant uptake between organic and inorganic forms (Gracey and Stewart 1974).

The absorption and accumulation of mercury in plants varies with plant parts and species. Comparatively, higher levels are accumulated in roots than other plant parts (Hogg *et al.* 1978; Lenka *et al.* 1992). Translocation from the roots to various plant tissues takes place, with concentrations decreasing in the following order: roots > stems > leaves > seeds (Talmage and Walton 1993). Plant tissue concentrations ranging from 0.01 to 2.0 mg $Hg \cdot kg^{-1}$ at a contaminated site averaging 269 mg $Hg \cdot kg^{-1}$ soil and soil to plant coefficient ratios ranging from 3.7×10^{-5} for seeds to 7×10^{-3} for grass blades have been reported by Talmage and Walton (1993).

The background levels of mercury in vegetables and fruits vary from 2.6 to 86 $\mu g \cdot kg^{-1}$ dw and 0.6 to 70 $\mu g \cdot kg^{-1}$ fw, respectively, (Kabata-Pendias and Pendias 1992). Lindqvist (1992) and Kabata-Pendias and Pendias (1992) reported that the total contents of mercury in vegetation are usually below 100 $\mu g \cdot kg^{-1}$ dw.

In the Canadian environment, Gracey and Stewart (1974) reported plant contents of 5 to 17 (mean 12) $\mu g\ Hg \cdot kg^{-1}$ for barley, of 4 to 19 (mean 9) $\mu g\ Hg \cdot kg^{-1}$ for oats, of 7 to 15 (mean 11) $\mu g\ Hg \cdot kg^{-1}$ for wheat, and of 39 $\mu g\ Hg \cdot kg$ for alfalfa. Plants grown at contaminated sites may accumulate much higher than normal amounts of mercury. Temple and Linzon (1977) observed 0.1 $mg\ Hg \cdot kg^{-1}$ in lettuce leaves grown in soils around chlor-alkali chemical works. Hogg *et al.* (1978) reported 0.09 to 2.01 $\mu g\ Hg \cdot kg^{-1}$ in bromegrass tops grown in sludge treated soils. In soils treated with fungicides and mercury salts, MacLean (1974) observed 0.36 to 0.44 and 0.11 to 0.32 mg $Hg \cdot kg^{-1}$ dw in potato and lettuce leaves, respectively. Weaver *et al.* (1984) reported 1.4 to 37.6, 0.4 to 1.2 and 80 to 800 mg $Hg \cdot kg^{-1}$ in leaves, stems, and roots, respectively, in bermuda grass (*Cynodon dactylon L*) grown on a contaminated soil with 50 mg $Hg \cdot kg^{-1}$.

Toxicity

A summary of available information on the effects of mercury on terrestrial plants is presented in Tables 8 and 9. Table 8 regroups the toxicity studies selected for use in soil quality guidelines derivation whereas table 9 presents other consulted studies.

Common symptoms of mercury toxicity to plants are inhibition of photosynthesis, stunted roots and stunted seedlings, all with consequent reductions in yield. Some studies reported that the accumulation of mercury in roots inhibits uptake of other elements, such as potassium (Kabata-Pendias and Pendias 1992).

Environment Canada (1995) studied toxicity of mercury on seedling emergence and root elongation in radish (*Raphanus sativa*) and *lettuce (Lactuca sativa*). Seedling emergence studies were conducted in 60 g artificial soils. Tests were conducted according to Protocol for Short Term Toxicity Screening of Hazardous Waste Sites, EPA 600/3-88-029 (Green *et al.* 1989). The NOEC, LOEC, EC_{25} and EC_{50} values on seedling emergence were 51, 103, 73 and 103 mg $Hg \cdot kg^{-1}$ for radish (72 h) and 7, 12, 11 and 15 mg $Hg \cdot kg^{-1}$ for lettuce (120 h), respectively. The LOEC for radish and lettuce corresponded to 52 and 27%, respectively, of seedlings not emerging. Root elongation studies were conducted with nutrient solutions using petri dishes. NOEC, LOEC, EC_{25}, and EC_{50} toxicity values of 12, 16, 15, and 70 mg $Hg \cdot L^{-1}$ for radish, and 6, 12, 9, and 18 mg $Hg \cdot L^{-1}$ for lettuce have been reported, respectively (Environment Canada 1995).

Beauford *et al.* (1977) reported that the lowest NOEC and LOEC for *Pisum sativum* in solution culture were 1 and 5 mg $Hg \cdot L^{-1}$, respectively. Toxic effects in young barley were reported at a mercury level of 3 $mg \cdot kg^{-1}$ soil dw (Davis *et al.* 1978; Dmowski and Karolewski 1979). Macnicol and Beckett (1985) established the critical level of mercury tissue content leading to depression of yield, between 1 and 8 $mg \cdot kg^{-1}$ dw of tissue for barley, cabbage, maize and oats.

Sheppard *et al.* (1993) observed significant detrimental effects, such as reduced or early bloom initiation in *Brassica rapa*, at 10 to 22 mg Hg·kg^{-1} in sand and at 220 mg Hg·kg^{-1} in clay and garden soils. They reported EC_{50} values for seedling emergence, first bloom, plant height and stem dry weight in turnip (*Brassica rapa*) as beeing 50, 7, 70 and 40, respectively, in sand, whereas in clay and garden soils these values were much higher, ranging from 550 to 1000 mg Hg·kg^{-1}. Regarding seedling emergence in lettuce, the EC_{50} values were 30, 200 and 500 mg·kg^{-1} in sand, clay and garden soils, respectively (Sheppard 1993). Based on these bioassays a toxicity threshold lying between 2 and 20 mg Hg·kg^{-1} soil was suggested for inorganic mercury (Sheppard 1994).

Reduction in bermudagrass (*Cynodon dactylon*) growth was observed in silt loam and fine sand soils containing 8 and 50 mg Hg·kg^{-1}, respectively but no effect was observed in black clay at 50 mg Hg·kg^{-1} (Weaver *et al.* 1984). Soils were spiked with $HgCl_2$ and the plants, grown in pots, were analyzed after 6 weeks. These results suggest that increased clay content of soil reduces the toxicity of mercury to plants. On the other hand, Estes *et al.* (1973) observed no toxic effects from high concentrations of mercury in soil (455 mg·kg^{-1}) on bent grass (*Agrostis canina*) from a golf course green and relate this to the fact that the top soil had been enriched in organic matter.

The vapour from metallic mercury has been demonstrated to be harmful to plants (EPS 1979). Therefore, the phytotoxicity of mercury compounds can also be due to mercury vapours arising from thermal decomposition or catalytic reductions to metallic vapour. Some of the mercury fungicides, such as DPM (diphenylmercury) are decomposed in soils at room temperature to give metallic mercury vapour (EPS 1979).

The toxicity of volatilized elemental mercury and some methylated compounds are known to be the most serious for plants. Siegel *et al.* (1984) suggested that the most active toxicant is elemental mercury, not ionic mercury. They reported NOEC of 14 µg Hg vapour·L^{-1} on seed germination and early seedling growth in several plant species and suggested that young plants are more sensitive to mercury saturated air than mature plants.

4.3 Terrestrial Invertebrates

Uptake Metabolism and Elimination

Mercury accumulates in soil invertebrates even at low soil concentrations (Rundgren *et al.* 1992). Mercury concentrations in invertebrates ranging from 0.79 mg·kg^{-1} for harvestman (*Phalangida*), to 15.5 mg·kg^{-1} for earthworms (*Oligochaeta*) have been reported by Talmage and Walton (1993) in a mercury contaminated site. Mean food chain transfer coefficients, defined as the ratio of mercury concentration in the whole body to mercury concentration in food, were 0.88 for herbivore/omnivore invertebrates and 2.35 for carnivore invertebrates.

Beyer *et al.* (1985) reported that the methylmercury accumulated in earthworms was up to 22 times greater than the soil concentration. Concentrations found in *Eisenia fetida* were 13, 27, and 85 mg Hg·kg^{-1} ww, whereas the potting soil in which they were kept for 12 weeks contained 0.63, 1.3 and

3.8 mg·kg^{-1} soil ww respectively (soil water content was 60%). Bull *et al.* (1977) reported that the earthworms (*Lumbricus terrestris*) in a contaminated soil (3.81 mg Hg·kg^{-1}) collected near a chlor-alkali works, contained higher concentrations of Hg (1.29 mg Hg·kg^{-1} ww) than those at 10 to 30 km from the site (0.106 mg Hg·kg^{-1} dw and 0.041 mg Hg·kg^{-1} ww for soil and worms respectively). Helmke *et al.* (1979) observed concentrations ranging from 0.5 to 0.66 mg Hg·kg^{-1} dw in earthworms living in soil containing less than 0.02 mg Hg·kg^{-1}. Siegel *et al.* (1975) reported 0.39 mg Hg·kg^{-1} dw in earthworms from soil containing 0.014 mg Hg·kg^{-1}.

Toxicity

A summary of available information on the effects of mercury to terrestrial invertebrates is presented in Tables 8 and 10. Table 8 regroups the toxicity studies selected for use in soil quality guidelines derivation whereas table 10 presents other consulted studies.

Environment Canada (1995) determined NOEC, LOEC, LC_{25} and LC_{50} values for the survival of earthworms (*Eisenia fetida*) exposed to $HgCl_2$ in artificial soils. Tests were conducted according to the Protocol for Short Term Toxicity Screening of Hazardous Waste Sites, EPA 600/3-88-0290 (Green *et al.* 1989). They reported concentrations of 96, 194, 130 and 181 mg Hg·kg^{-1} soil for NOEC, LOEC, LC_{25} and LC_{50}, respectively. The LOEC corresponded to 58% mortality and was therefore higher than LC_{25} and LC_{50}values.

The toxicity of mercury to terrestrial invertebrates varies with the type of soil and the organism. Sheppard *et al.* (1993) reported LD_{50} values of 60 and 700 mg Hg·kg^{-1} in sand and garden soils, respectively, on mortality of earthworms (*Lumbricus terrestris*). Fisher and Koszorus (1992) observed NOEC, LC_{25} and LC_{50} at 100, 250 and 300 mg Hg·kg^{-1}, respectively, on mortality of earthworms (*Eisenia fetida*) in a mixture of peaty marshland soil and horse manure.

Marigomez *et al.* (1986) observed that, following the administration of 0 to 1000 mg Hg·kg^{-1} of feed to the terrestrial slug (*Arion ater*) for 27 days, food consumption decreased significantly in a dose-dependent manner, at exposure levels greater than 10 mg Hg·kg^{-1} diet. Growth was significantly disrupted at the highest dose, 1000 mg Hg·kg^{-1} feed, and was probably due to decreased food consumption.

The toxicity of mercury compounds to earthworms exposed to contaminated soil or sludge is variable. Abbasi and Soni (1983) exposed earthworms, *Octochaetus pattoni*, to mercuric chloride for 60 days at levels of 0 to 5.0 mg Hg·kg^{-1} in soil mixed with animal dung. LC_{50} estimates ranged from >5 mg Hg·kg^{-1} at 5 days to 0.79 mg Hg·kg^{-1} at 60 days.

Beyer *et al.* (1985) measured the survival of earthworms exposed for a 12 week period to soil containing MMC (CH_3HgCl) at 0 to 125 mg·kg^{-1} ww. None of the earthworms survived 12 weeks at 25 and 125 mg MMC·kg^{-1}. The reported NOEC and LOEC values corresponded to 3.25 and 9.5 mg Hg·kg^{-1} soil dw. The results also showed that the chronic toxicity of mercury compounds, especially methylmercuric chloride, is lower in *E. fetida* as compared to *O. pattoni*. Haney and

Lipsey (1973) reported EC_{31}, EC_{41} and EC_{100} values of 0.004, 0.02 and 0.04 mg $Hg \cdot L^{-1}$, respectively, for reproduction of aphids grown on tomato plants treated with mercury in nutrient solution.

4.4 Mammals and Birds

Uptake, Metabolism and Elimination

The most frequent exposure route of mercury contamination for domestic animals is through the consumption of seeds treated with mercury fungicides and to a lesser extent, inhalation of elemental mercury.

The absorption of methylmercury through the mammalian gastrointestinal tract is generally high (more than 90%) and indirect evidence suggests that ethyl, propyl, butyl, amyl, and hexyl mercury compounds have similar absorption rates. Phenylmercuric acetate absorption is lower (40-50% of oral dose) (NRCC 1979).

The appreciable lipid solubility of metallic mercury vapour is responsible for the high rate of absorption in the lungs (WHO 1976). The gastrointestinal absorption of liquid metallic mercury is extremely low and has been estimated at 0.01% (WHO 1976). The percutaneous absorption of MMD in guinea pigs, after 5 hours of exposure was approximately 6% of the applied mercury (WHO 1976).

Mercury levels are generally highest in the kidney and liver tissues (NRCC 1979). Mercury, in vapor and organic forms, is also readily absorbed and stored in brain tissue (Aschner and Aschner, 1990). Nielsen *et al.* (1991) reported a kidney concentration of 47.4 mg $Hg \cdot kg^{-1}$ after two days when mice were administered a single dose of 44 mg $Hg \cdot kg^{-1}$ body weight. Talmage and Walton (1993) reported mean mercury concentrations in kidney tissue of 1.16 and 38.8 $mg \cdot kg^{-1}$ for mouse (*Peromyscus leucopus*) and shrew (*Blarina brevicauda*), respectively, at a contaminated site. They reported transfer coefficients for diet to kidney of 0.75 and 4.4 for *P. leucopus* and *B. brevicauda*, respectively.

The principal routes of elimination of mercury from the body are through the urine and feces. For inorganic mercury, the faecal route is dominant soon after exposure whereas the urinary route is favoured in cases of high doses or prolonged exposure (WHO 1976). For methyl mercury, the faecal route is the most important in the elimination of mercury after acute or chronic dosing (NRCC 1979; WHO 1976). Howe *et al.* (1972) found that approximately 90% of a 1 $mg \cdot kg^{-1}$ dose of mercuric chloride fed to lactating goats was recovered in the feces. Only 3 to 4% was recovered in the urine.

The half-life of methylmercury in mammals varies considerably with species and dose administered. Half-life values of 6 to 7 days in mice, 40 to 51 days in rats, 76 days in cats and 150 days in monkeys have been reported (NRCC 1979). Goyer (1991) reported a half-life of 40 days for inorganic mercury in mammals.

Commercial mercury compounds such as Panogen 15 (2.5% methylmercurydicyanide) are used in Canada in seed treatment (Fimreite 1970). It has been demonstrated that seed-eating birds such as

horned larks (*Eremophila alpestris*), partridges (*Perdix perdix*), pheasants (*Phasianus colchicus*), and predators like prairie falcons (*Falco mexicanus*) collected in the grain-growing districts of Alberta and Saskatchewan, frequently carry elevated levels of mercury (Fimreite 1970).

As for mammals, the highest concentrations of mercury in exposed birds are generally found in liver and kidney tissues. Also similarly for birds and mammals, methylmercury has a higher rate of absorption and a longer half-life than inorganic mercury (WHO 1989).

Bioaccumulation

Following chronic oral exposures of pigs to phenylmercuric chloride, at doses ranging from 0.19 to 4.56 mg $Hg \cdot kg^{-1}$ bw, residue levels were highest in kidney (40 to 200 $mg \cdot kg^{-1}$) and liver (5 to 72 $mg \cdot kg^{-1}$) tissues (Tryphonas and Nielsen 1973). Mercury residues in the intestine ranged from 4 to 34 $mg \cdot kg^{-1}$, while levels detected in the muscle and brain tissues were much lower.

Wright *et al.* (1973) fed sheep and cattle with 0.33 $mg \cdot kg^{-1}$ bw of MMD per day for 42 to 60 days. Residue levels were highest in kidney tissue (110 to 165 mg $Hg \cdot kg^{-1}$) followed by liver (40 to 54 mg $Hg \cdot kg^{-1}$), muscle (14 to 23 mg $Hg \cdot kg^{-1}$) and brain (12 to 13 mg $Hg \cdot kg^{-1}$) tissues.

Although there has been some reports of mercury poisoning and contamination in wild mammals due to mercury treated seeds (Jefferies *et al.* 1973), the concentration of mercury in tissues of terrestrial herbivores is typically very low. Several studies have reported low mercury levels in different species of deer, moose and caribou (Desai-Greenaway and Price 1976; Kocan *et al.* 1980). Talmage and Walton (1993) reported mercury concentrations in the following order: herbivores < carnivores < detritivores.

Frank *et al.* (1979) analyzed mercury levels in six species of wild animals in southern Ontario. They observed higher levels of mercury (0.4 to 0.95 $mg \cdot kg^{-1}$) in liver than in brain and muscle tissues (0.05 to 0.099 $mg \cdot kg^{-1}$) for red fox (*Vulpes fulva*), racoon (*Procyon lotor*) and striped skunk (*Mephitis mephitis*). But for fisher (*Martes pennanti*), marten (*Martes americana*), and mink (*Mustela vison*), they found higher levels (0.33 to 0.71 $mg \cdot kg^{-1}$) of mercury in muscle tissues. Smith and Armstrong (1976) reported that the concentration of mercury in terrestrial carnivores such as coyotes, foxes and wolves was generally low (< 0.20 $mg \cdot kg^{-1}$) in uncontaminated areas. These studies suggest that the mesured level of bioaccumulation of mercury in terrestrial mammals varies with species and organs sampled.

Mercury concentrations are generally low in tissues of semi-aquatic herbivorous mammals such as muskrats (*Ondatra zibethicus*) and beavers (*Castor canadensis*). Muscle and liver mercury levels in these species are generally lower than 0.10 $mg \cdot kg^{-1}$ (Radvanyi and Shaw 1981; Wren 1984). Although most studies suggest that mercury levels in beavers and muskrats are independent of environmental contamination, Desai-Greenway and Price (1976) reported that mercury levels in muskrat tissues in the St. Clair River, Ontario, decreased (from 0.42 to less than 0.01 $mg \cdot kg^{-1}$) between 1969 and 1976 in response to curtailed discharge of mercury into the river.

Tissue mercury levels of semi-aquatic omnivores and piscivores such as raccoons (*Procyn lotor*), minks *(Mustela vison*) and otters (*Lutra canadensis*) are typically elevated in comparison to herbivores, even in uncontaminated regions (Wren *et al.* 1988). Liver mercury levels in otter from uncontaminated regions of Ontario are in the range of 1.5 to 4.0 $mg \cdot kg^{-1}$ (Wren *et al.* 1988). Cases of mercury poisoning in wild mink and otter have been documented in Canada near point source discharges of mercury in waterways (Wren 1986).

Finley and Stendell (1978) fed black ducks a diet containing 3 mg $Hg \cdot kg^{-1}$ feed (as MMD) for 28 weeks. Mercury levels were highest in adult bird feathers, followed by liver and kidney tissues (61, 22, and 14 $mg \cdot kg^{-1}$ ww, respectively). Borg *et al.* (1970) fed goshawks a diet of contaminated chicken muscle and liver (average dietary mercury content 13 $mg \cdot kg^{-1}$). Three goshawks receiving chicken muscle and liver as their sole food died at 30, 38, and 47 days, while another one fed solely with contaminated chicken muscle (10 $mg \cdot kg^{-1}$) died within 39 days. Tissue methylmercury levels were high in liver, kidneys (112 and 82 $mg \cdot kg^{-1}$, respectively), skeletal muscles (38 to 51 $mg \cdot kg^{-1}$) and brain (35 to 45 $mg \cdot kg^{-1}$). The reproductive organs accumulated mercury to a maximum of 280 $mg \cdot kg^{-1}$ (Borg *et al.* 1970).

Toxicity

A summary of the toxicological data on the effects of mercury to mammals and birds is presented in Tables 11, 12, 13 and 14. Table 11 regroups the toxicity studies selected for use in soil quality guidelines derivation whereas tables 12, 13, and 14 present other consulted studies. Additional information can be found in Osweiler *et al.* (1985) and Venugopal and Luckey (1978).

Mercury is ubiquitous in the environment and usually present in animal tissues at low concentrations. Inorganic mercury is relatively biologically inert, but the addition of an organic group to form methylmercury (CH_3Hg) markedly increases its toxicity. There are reports, however, showing that inorganic mercury compounds produce fetotoxic effects including immune and pathological changes in the kidneys of rats and mice (Hultman and Enestrom 1992). Organic mercury is highly lipid soluble and readily transferred across biological membranes therefore posing a greater threat to animal health.

Mercury binds to protein-thiol groups causing severe cellular damage to the central nervous system (NRCC 1979). Typical clinical signs of methylmercury intoxications include anorexia, ataxia, irregular muscle movement and convulsions. Buck *et al.* (1973) observed similar symptoms for inorganic, aryl, and methoxyethyl forms of mercury poisoning in cattle. They also reported toxicosis of the central nervous system (CNS) in cattle fed with methylmercury. This symptom had been associated with methylmercury poisoning in other species.

Verschuuren *et al.* (1976b) fed weanling rats MeHgCl at dietary levels between 0 and 2 mg $Hg \cdot kg^{-1}$ body weight, for 2 years. The non-toxic effect level indicated by the 12 week study was 0.4 mg $Hg \cdot kg^{-1}$ body weight which induced a mercury level of 1.7 $mg \cdot kg^{-1}$ in liver tissues. At 2 mg $Hg \cdot kg^{-1}$ body weight, growth reduction, kidney damage and histochemical changes were observed. The mortality was higher in females (48%) and than in males (37%). In a 2 weeks study, Verschuuren *et al.* (1976a) reported central nervous system toxicity, weight loss and high mortality at 250 $mg \cdot kg^{-1}$

bw of MeHgCl. In a 12 week experiment, toxic signs, weight loss and restricted food intake were observed at 20 mg Hg·kg^{-1} from week 9 onwards (Verschuuren *et al.* 1976a).

Fimreite (1970) exposed 2 weeks old leghorn cockerels to a diet dosed with Panogen 15 (2.5% methyl mercury dicyandiamide) at concentrations of 6, 12, and 18 mg MMD·kg^{-1} for 3 weeks. All treated birds showed significant reductions in weight and a significant increase in deaths was observed at the highest dose.

Fimreite and Karstad (1971) used chicks fed with MMD (3.9, 7.2, and 10 mg Hg·kg^{-1} bw) as food for red-tailed hawks over a period of 12 weeks. All the poisoned hawks showed neurological symptoms, weakness, and impaired coordination of muscular movement when fed with the chicks of the 10 mg Hg·kg^{-1} group (20 mg Hg·kg^{-1} chick liver) .

Piper *et al.* (1971) fed pigs MMD in single acute oral doses between 0 and 107.4 mg Hg·kg^{-1} bw. The NOEC was observed at 1.7 mg Hg·kg^{-1}(0.2 mg Hg·kg^{-1} in spinal cord tissues). Pigs given 3.4 to 6.7 mg Hg·kg^{-1} showed various symptoms (CNS depression, loss of weight, weakness, vomiting staggering and stiffness) and then recovered. Pigs given 10 to 26.9 mg Hg·kg^{-1} showed CNS depression, anorexia, fever, and abnormal posturing. Massive exposure (54 to 107 mg Hg·kg^{-1}) resulted in diarrhea, vomiting, tachycardia, cyanosis, labored respiration, coma and death within 12 to 24 hours. These authors suggested a minimum acute lethal oral dose of 13.4 mg Hg·kg^{-1} bw. Three out of four pigs given doses equal or higher than 13.4 mg Hg·kg^{-1} bw died from methylmercurialism or were euthanatized in extremis.

Bhatnagar *et al.* (1982) fed one week old Peking ducks (*Anas platyrhynchos*) with a diet containing MMC at 0.03 to 13.43 mg Hg·kg^{-1} bw for 12 weeks. Ducks treated with 13.43 mg·kg^{-1} of MMC (observed brain tissue concentrations of 23.4 and 19.57 mg·kg^{-1} in female and male ducks, respectively) showed severe neurotoxicity and liver injury. Ducks treated with 3.76 mg·kg^{-1} showed a tendency to reduce weight.

Finley and Stendell (1978) fed black ducks (*Anas rubripes*) with a morsodren diet (2.2% of MMD) containing 3 mg Hg·kg^{-1} for 28 weeks during two consecutive breading seasons. Ducks fed with mercury produced only 16 ducklings that survived 1 week compared with 73 ducklings in controls. Whole embryos that failed to hatch had mercury concentration means of 9.62 and 6.08 mg·kg^{-1} during the first and second years, respectively. Brains of dead ducklings contained between 3.25 and 6.98 mg Hg·kg^{-1} and exhibited lesions characteristic of mercury poisoning.

Tryphonas and Nielsen (1973) fed 5 weeks old pigs with MMD and EMC at 0 to 0.76 mg Hg·kg^{-1} bw for 60 to 90 days. Pigs fed 0.38 and 0.76 mg Hg·kg^{-1} body weight developed a progressive cerebral deficiency after 20 to 90 days. The results indicated that brain tissue concentrations of 14 mg Hg·kg^{-1} (obtained with an oral daily dose of 0.38 mg Hg·kg^{-1} bw) were always associated with clinical signs and severe lesions, while brain tissue concentrations below 7 mg Hg·kg^{-1} (obtained with an oral daily dose of 0.19 mg Hg·kg^{-1} bw) were difficult to correlate with lesions.

Heinz (1974) fed mallard ducks a diet containing MMD (0.5 and 3.0 mg $Hg \cdot kg^{-1}$ bw) for 21 weeks. The lower dose had no effect on reproduction but 3.0 mg $Hg \cdot kg^{-1}$ reduced egg laying and increased embryonic and duckling mortality. Heinz (1976) fed mallards for 2 consecutive years with 0.5 mg $Hg \cdot kg^{-1}$ bw of MMD and found no significant effect on egg production, hatching and behaviour of ducklings. Häegele *et al.* (1974) observed no significant effects on eggshell thickness of female ducks dosed with 200 $mg \cdot kg^{-1}$ of Ceresan M. (3.1% ethylmercury).

Spann *et al.* (1986) fed twelve-day old bobwhite chicks with a diet containing 0, 5.4 and 20 $mg \cdot kg^{-1}$ methylmercury chloride for 6 weeks and reported 80% mortality at 20 mg $MM \cdot kg^{-1}$ (16 mg $Hg \cdot kg^{-1}$). Hill and Schaffner (1976) fed Japanese quail from hatching to one year of age on diets containing mercuric chloride salt (from 0 to 32 mg $Hg \cdot kg^{-1}$). Food consumption, growth rate, weight maintenance, hatchability, and eggshell thickness were unaffected. However, the rate of egg fertilization was generally depressed in birds fed with mercury levels higher than 4 $mg \cdot kg^{-1}$. In contrast to short term (5 d) experiments, chronic exposure to organic mercury compounds was shown to affect reproduction in birds at dietary concentrations as low as 3 mg $Hg \cdot kg^{-1}$ (Finley and Stendell 1978).

Scheuhammer (1988) fed Zebra finches *(Poephila guttata)* with diet containing 0, 0.35, 0.88 and 1.75 $mg \cdot kg^{-1}$ $bw.day^{-1}$ for 76 days. He reported a NOEC of 0.88 and 25% mortality at 1.75 $mg \cdot kg^{-1}$ $bw.day^{-1}$. Mullins *et al.* (1977) reported LD_{50} of 65 $mg \cdot kg^{-1}$ bw on pheasant (*Phasianus colchicus*) mortality in subacute studies

Nicholson and Osborn (1984) observed kidney lesions in juvenile starlings (*Sturnus vulgaris*) fed with a commercial diet containing mercury (1.1 mg $Hg \cdot kg^{-1}$). No signs of overt toxicity were seen in the birds. Damage to the kidney was mainly confined to the proximal tubules, and was similar to that found in mercury-contaminated sea birds in the field.

5. DERIVATION OF ENVIRONMENTAL SOIL QUALITY GUIDELINES

5.1 Introduction

Canadian soil quality guidelines are designed to protect four different land uses: agricultural, residential/parkland, commercial and industrial. The Canadian Soil Quality Guidelines for mercury are based on the procedures described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996).

All data selected for use in the following derivations have been screened for ecological relevance and are presented in Tables 6, 8 and 11. All of the toxicological studies consulted, including those not used for guideline derivation, are presented in Tables 6 to 14. Studies may be excluded from use because of one or more of the following reasons:

- soil pH was not recorded or was below 4 (which is outside pH range of most Canadian soils)
- no indication of soil texture was provided

- inappropriate statistical analysis was used
- test was not conducted using soil or artificial soil
- test soil was amended with sewage sludge or a mixture of toxicants
- test did not use controls

LOEC and EC data used in the following derivations were considered to be biologically significant in addition to statistically significant according to the study from which the data were taken.

According to Section 7.5.2.2 of the Protocol, the geometric mean must be used when multiple data are available for the same endpoint with the same species. For the mercury data, the geometric mean has not been applied.

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

5.2.1 Soil Quality Guideline for Soil Contact (SQG_{SC})

The derivation of the SQG_{SC} is based on toxicological data for vascular plants and soil invertebrates. The toxicological data for plants, and invertebrate, selected according to CCME (1996), are presented in Table 8. There were sufficient toxicological data to use the preferred weight of evidence method for guideline derivation.

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

$$TEC = NPER/UF$$

where,

TEC = threshold effects concentration ($mg \cdot kg^{-1}$ soil)

NPER = no potential effects range (25th percentile of effects and no effects data distribution) ($mg \cdot kg^{-1}$ soil)

UF = uncertainty factor (if needed); an uncertainty factor of 4 was applied because more than 25 % data below 25th percentile are definitive effects and more than 50 % of the data are nominal concentrations.

Out of a total of 31 data points, the 25th percentile corresponds to the 8th datum point of 40 $mg \cdot kg^{-1}$ soil from the Sheppard et al (1993) study on turnips.

Thus,

TEC = 40/4 = 10 $mg \cdot kg^{-1}$ soil

Nutrient and Energy Cycling Check

The nutrient and energy cycling check for the SQG_{SC} for agricultural, residential/parkland land uses was calculated using the selected microbial processes data presented in Table 6. Nitrification and

nitrogen fixation data are considered to be primary data, whereas nitrogen mineralisation, denitrification and carbon cycling data are considered secondary data. LOEC data, as reported by the author are used directly while effective concentration (EC) data producing >15 and < 50% effects in primary data (i.e. EC_{15} to EC_{50}) and >15 and < 35% effects in secondary data (i.e. EC_{15} to EC_{35}) are interpreted as LOEC values. Only secondary data were available and the check was carried out using a modified LOEC method whereby the geometric mean of available LOECs is calculated as the nutrient and energy cycling check.

The nutrient and energy cycling check (NECC) was calculated as follows.

$$NECC = (LOEC_1 \bullet LOEC_2 \bullet LOEC_3 ... LOEC_n)^{1/n}$$

where,
NECC = effects concentration low ($mg \cdot kg^{-1}$ soil)
LOEC = lowest observed effects concentration ($mg \cdot kg^{-1}$ soil)
n = number of available LOECs

Thus,
NECC $= (0.1 \cdot 10 \cdot 50 \cdot 74 \cdot 200)^{1/5} = 15\ mg \cdot kg^{-1}$ soil

Since the TEC (10 $mg \cdot kg^{-1}$ soil) is lower than the NECC (15 $mg \cdot kg^{-1}$ soil) the TEC is considered to be protective of microbial nutrient and energy cycling processes and is adopted directly as the SQG_{SC} for agricultural, residential/parkland land uses.

5.2.2 Soil Quality Guidelines for Dry Matter Ingestion (SQG_I)

The soil quality guideline for ingestion applies only to agricultural land use.

Calculation of the SQG_I is based on the lowest observed adverse effects level (LOAEL) taken from the selected mammalian and avian toxicological data listed in Table 11. The lowest observed adverse effects level, indicating the species most threatened was 0.38 $mg \cdot kg^{-1}$ $bw \cdot day^{-1}$ which resulted in clinical toxicological signs and lesions in pigs (Tryphonas and Nielsen 1973).

The LOAEL is used to calculate the daily threshold effects dose (DTED) according to the equation:

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

where,
DTED = daily threshold effects dose ($mg \cdot kg^{-1}$ $bw \cdot day^{-1}$)
LOAEL = lowest observed adverse effects dose ($mg \cdot kg^{-1}$ $bw \cdot day^{-1}$)
UF = uncertainty factor; an uncertainty factor of 2 was applied as the LOAEL was considered to be biologically significant and extrapolation below this level is required.

Thus,
DTED = 0.38/2 = 0.19 $mg \cdot kg^{-1}$ $bw \cdot day^{-1}$

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% of the total exposure (CCME 1996), the remaining 75% 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% of the DTED.

$$\text{exposure from direct soil ingestion} + \text{exposure from food ingestion} = 0.75 \cdot \text{DTED}$$

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:

$$\text{SIR} = \text{DMIR} \bullet \text{PSI}$$

where,

SIR = the soil ingestion rate (kg dw soil·day^{-1})

DMIR = geometric mean of available dry matter intake rates (kg·day^{-1}); estimated to be 3 kg·day^{-1} (Warrington 1995).

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 = $3 \bullet 0.083 = 0.249$ kg dw soil·day^{-1}

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% of the DTED when combined with the exposure calculated for food ingestion.

$$\text{exposure from soil ingestion} = \text{SIR} \cdot \text{BF} \cdot SQG_I / \text{BW}$$

where,

SIR = soil ingestion rate (kg dw soil·day^{-1})

BF = bioavailability factor; Due to lack of specific information on the bioavailability of mercury from ingested soil for live stock and terrestrial wildlife, a bioavailability factor (BF) of 1 is assumed (CCME 1996).

SQG_I = concentration of the contaminant in soil that will not result in greater than 75% DTED

BW = mean body weight (kg); the mean body weight of pigs was determined to be 80 kg (Warrington 1995).

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^{-1})

DMIR = geometric mean of dry matter intake rates (kg dw food·day^{-1});estimated to be 3 kg·day^{-1} (Warrington 1995).

SIR = soil ingestion rate (kg dw soil·day^{-1})

Thus,

FIR = 3 - 0.249 = 2.75 kg dw food·day^{-1}

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

$$\text{exposure from food ingestion} = FIR \cdot BCF \cdot SQG_I / BW$$

where,

FIR = food ingestion rate (kg dw food·day^{-1})

BCF = bioconcentration factor; calculated from the data on plant accumulation of mercury presented in Table 15 using the CCME (1994) procedure and determined to be 0.125.

SQG_I = concentration of the contaminant in soil that will not result in greater than 75% DTED

BW = mean body weight (kg); the mean body weight of pigs was determined to be 80 kg (Warrington 1995).

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.

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

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

$$SQG_I = (0.75 \cdot 0.19 \cdot 80) / (0.249 \cdot 1 + 2.75 \cdot 0.125)$$

$$SQG_I = 19\ mg\ Hg \cdot kg^{-1}\ soil$$

5.3 Soil Quality Guidelines for Commercial and Industrial Land Uses

5.3.1 Soil Quality Guidelines for Soil Contact (SQG_{SC})

The derivation of the SQG_{SC} is also based on toxicological data for vascular plants and soil invertebrates presented in Table 8. However, for commercial and industrial land uses only the effects data are used and uncertainty factors are not applied. There was sufficient toxicological data to use the preferred weight of evidence method for guideline derivation.

The effects concentration low (ECL) is calculated as:

$$ECL = ERL$$

where,
ECL = effects concentration low ($mg \cdot kg^{-1}$soil)
ERL = effects range low (25th percentile of effects data distribution) ($mg \cdot kg^{-1}$)

Out of a total of 25 data points, the 25th percentile corresponds to the 6th datum point of 30 $mg \cdot kg^{-1}$ soil from the Sheppard et al. (1993) study on lettuce seedling emergence.
Thus,
ECL = 30 $mg \cdot kg^{-1}$ soil

Nutrient and Energy Cycling Check

The nutrient and energy cycling check for the SQG_{SC} for commercial and industrial land uses was also calculated using the selected microbial processes data presented in Table 12. Nitrification and nitrogen fixation data are considered to be primary data, whereas nitrogen mineralisation, denitrification and carbon cycling data are considered secondary data. LOEC data, as reported by the author are used directly while effective concentration (EC) data producing >15 and < 50% effects in primary data (i.e. EC_{15} to EC_{50}) and >15 and < 35% effects in secondary data (i.e. EC_{15} to EC_{35}) are interpreted as LOEC values. Only secondary data were available and the check was carried out using a modified LOEC method whereby the geometric mean of available LOECs is calculated as the nutrient and energy cycling check.

The nutrient and energy cycling check (NECC) was calculated as follows.

$$NECC = (LOEC_1 \bullet LOEC_2 \bullet LOEC_3 ... LOEC_n)^{1/n}$$

where,
NECC = effects concentration low ($mg \cdot kg^{-1}$ soil)
LOEC = lowest observed effects concentration ($mg \cdot kg^{-1}$ soil)
n = number of available LOECs
Thus,
NECC = $(0.1 \cdot 10 \cdot 50 \cdot 74 \cdot 140 \cdot 200 \cdot 1003 \cdot 1003)^{1/8} = 57$ $mg \cdot kg^{-1}$ soil

Since the ECL (30 $mg \cdot kg^{-1}$ soil) is lower than the NECC (57 $mg \cdot kg^{-1}$ soil) the ECL is considered to be protective of microbial nutrient and energy cycling processes and is adopted directly as the SQG_{SC} for commercial and industrial land uses.

5.4 Derivation of final Environmental Soil Quality Guidelines (SQG_E)

The following environmental effects-based soil quality guidelines for inorganic mercury should be used with soils within the pH range of 4 to 8.3. The toxicological studies upon which these guidelines are based were conducted within this pH range. Table 16 presents a summary of the derived soil quality guidelines for mercury and a comparison of the derived values with Ontario background levels and the CCME interim remediation criteria.

Agricultural land use

The lower value from the two procedures (SQG_{SC} and SQG_I) is selected as the final environmental effects-based soil quality guideline for agricultural lands (SQG_E). The lower of the two procedures is the SQG_{SC}. Therefore, the final SQG_E is 10 mg $Hg \cdot kg^{-1}$ soil.

Residential/Parkland uses

The SQG_{SC} of 10 mg $Hg \cdot kg^{-1}$ soil is the final SQG_E for residential/parkland uses.

Commercial and Industrial land uses

The SQG_{SC} of 30 mg $Hg \cdot kg^{-1}$ soil is the final SQG_E for commercial and industrial land uses.

6. DATA GAPS

Sufficient data exist in the literature on the fate and behaviour of mercury in the terrestrial environment. However, very few appropriate data exist on toxicity of mercury to soil terrestrial receptors. Additional information is also needed on regional background levels of mercury and bioavailability in soils across Canada.

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TABLES

Table 1. Physical and chemical properties of some mercury compounds

Property	Elemental mercury	Mercurous chloride Mercury (I) chloride	Mercuric chloride, Mercury (II)chloride	Methylmercury chloride	Dimethyl mercury
Empirical formula	Hg^0	Hg_2Cl_2	$HgCl_2$	CH_3HgCl	$(CH_3)_2Hg$
Molecular weight	200.59	472.09	271.49	251.09[1]	230.67
CAS registry number	7439-97-6	10112-91-1	7487-94-7	22967-92-6	593-74-8
Common synonyms	hydrargyrum, liquid silver, quicksilver	Calomel, mild mercury chloride, mercury monochloride, mercury protochloride, mercury subchloride	mercury bichloride, mercury perchloride, corrosive mercury chloride	NA	mercury, dimethyl, methyl mercury
Physical state (20°C)	heavy, mobile, liquid metal	heavy powder	crystals, granules or powder	crystalline	volatile liquid
Melting point (°C)[2,3]	-38.83	sublimes at 400-500	277	170	NA
Boiling point (°C, 1 Atm.)	356.72	NA	NA	NA	96
Density ($g \cdot cm^{-3}$ @ 25°C)	13.5	7.15	5.4	4.06	3.18 (20°C)
Electrical resistivity (μohm.m @ 20°C)	95.76	NA	NA	NA	NA
Octanol/water partition coefficient (K_{ow}, D)	NA	NA	0.81[4] 0.61[5]	2.0[6] 2.54[5]	NA
Log K_{ow}	NA	NA	-0.092[4] -0.215[5]	0.301[7] 0.405[6]	NA
Solubility in water ($mg \cdot L^{-1}$ 25°C)	0.002[7] (30°C)	2.00	6.9×10^4	1016[6]	insoluble
Solubility in benzene ($mg \cdot L^{-1}$)	2.387	insoluble	5000	6535[4]	NA
Solubility in alcohol ($mg \cdot L^{-1}$)	NA	insoluble	2.6×10^5	NA	soluble
Solubility in ether ($mg \cdot L^{-1}$)	NA	insoluble	4.6×10^4	NA	soluble

All data from Windholz *et al.* 1983 unless otherwise noted. 1 Weast and Astle (1982); 2 WHO (1989); 3 Environment Canada (1981); 4 Hantzsh and Vagt (1981); 5 Halbach (1985); 6 Eisler (1987); 7 Medeiros *et al.* 1980

Table 2. Mercury compounds and their general uses in Canada

Compound	Formula	Uses
Mercuric chloride	$HgCl_2$	Agriculture, medicine, photography
Mercuric cyanide	$Hg(CN)_2$	Medicine
Mercuric iodide	HgI_2	Medicine
Phenylmercuric acetate (PMA)	$C_6H_5HgOC(O)CH_3$	Agriculture, paints
Mercuric oxide	HgO	Agriculture, paints, batteries
Methylmercury dicyandiamide (MMD)	$C_3H_6HgN_4$	Agriculture
Mercuric sulphide (cinnabar)	HgS	Pigment
Methoxyethylmercuric acetate	$CH_3OC_2H_4HgOC(O)CH_3$	Pulp and paper, agriculture
Ethyl mercuric chloride	C_2H_5HgCl	Agriculture

(Adapted from Environment Canada 1981)

Table 3. Mercury emissions by sector

Sector	Total (kg)	Percent
Electrical Equipment Manufacturing	18	0.05
Gold Recovery	37	0.09
Paint Manufacturing	65	0.16
Battery Manufacturing	67	0.17
Dental Amalgams	85	0.21
Sewage Sludge Incineration	124	0.31
Pharmaceutical Use	236	0.59
Instrumentation	244	0.61
Thermometer Breakage	403	1.01
Natural Gas Combustion	517	1.30
Wood Combustion	767	1.92
Agricultural Chemicals	1103	2.77
Municipal Incineration	1623	4.07
Chlor-Alkali Industry	2374	5.96
Petroleum Combustion	2735	6.86
Fluorescent Tube Breakage	3375	8.47
Paint Application	4770	11.97
Coal Combustion	5042	12.65
Base Metal Recovery	16270	40.83
Total	39855	

(Environment Canada 1981)

Table 4. Mercury levels in Canadian soils

Location	Soil type	Concentration ($mg \cdot kg^{-1}$)	Range ($mg \cdot kg^{-1}$)	Reference
Arctic, temperate		0.07	0.02 - 0.15	Jonasson and Boyle 1972
Worldwide	Fossil fuels			
	-peat	--	0.06 - 0.3	Jonasson and Boyle 1972
	-coal	--	0.01 - 8.53	
	-coal in mercuriferous basins	--	0.02 - 300	
	-crude oil	--	0.02 - 2	
	-bitumens, asphalts, solid hydrocarbons	--	2 - 900	
Canada	Tills, glacial clay, sand	0.05	0.02 - 0.10	Jonasson and Boyle 1972
	Sandy soils	0.06	0.01 - 0.70	Frank *et al.* 1976
	Loam soils	0.09	0.02 - 0.78	Frank *et al.* 1976
	Gleysols	0.053	0.018-0.19	McKeague and Kloosterman 1974
	Organic soils	0.41	0.05 - 1.11	Frank et al. 1976
	General	0.06	0.005-0.10	McKeague and Wolynetz, 1980
	Soils in golf course greens	53	27 - 65	MacLean *et al.* 1973
Ontario	Soils in unmineralized belts			
	A horizon	0.16	0.06 - 0.20	Jonasson and Boyle 1971
	B horizon	0.09	0.03 - 0.14	
	C horizon	0.10	0.025 - 0.15	
	Soils in mineralized belts			
	A horizon	0.48	0.2 - 1.80	Jonasson and Boyle 1971
	B horizon	0.28	0.14 - 0.61	
	C horizon	0.26	0.15 - 0.55	
	-agricultural soils	0.11	0.01 - 1.14	Frank *et al.* 1976
		0.16	--	OMEE 1994
	-residential/parkland soils	0.23	--	
Saskatchewan	- cultivated soils	0.023	0.011-0.060	Gracey and Stewart 1974
	- uncultivated	0.022	0.005-0.057	
Alberta	- Southern Alberta soils	0.066	0.036 - 0.091	George *et al.* 1994
	- chernogems (A horizon)	0.026	0.02 - 0.035	Dudas and Pawluk 1976
	(C horizon)	0.044	0.030 - 0.070	
	- luvisols (A horizon)	0.011	0.010 - 0.012	
	- orthic gray, luvisols, black chernogems, humic gleysols (forest soils)	--	0.03 to 0.135	Dudas and Cannon 1983
British Columbia	- unmineralized areas	--	0.01 - 0.05	Environment Canada 1979
	- near mineralization	--	0.05 - 2.50	
	- mineralized areas	--	0.25 - 2.50	
Quebec	forest soils -surface horizons			
South Quebec		0.26	--	Grondin *et al.* (1994)
North Quebec		0.10	--	
Northen Manitoba	- A horizon	0.090	<0.005 - 0.220	Bodaly *et al.* 1984
	- C horizon	0.041	<0.005 - 0.180	

Table 5. Existing guidelines, criteria and recommended levels for mercury in soils

Jurisdiction	Category	Guideline ($mg \cdot kg^{-1}$)	Reference
Canada	Interim Assessment Criteria Interim Remediation Criteria (to protect human and environmental health)	0.1 (A) 0.8 (Agr); 2.0 (R/P); 10.0 (C/I)	CCME 1991
Alberta	assessment criteria	0.2	Alberta Environment 1990
Ontario	Clean-up guideline: Surface soil with potable groundwater situation (pH 5 to 9) Surface soil with non-potable groundwater situation (pH 5 to 9) Sub -surface soil with potable and non-potable groundwater situation (pH 5 to 11)	10 (Agr); 10 (R/P); 10 (C/I) 10 (R/P); 10 (C/I) 57 (R/P); 57 (C/I)	OMEE 1994
British Columbia	soil remediation criteria	0.1 (A); 2 (Agr, R/P); 10 (C/I)	BCMOE 1989
Quebec	Remediation guidelines	0.2 (A); 2 (B); 10 (C)	MENVIQ 1988
Japan	clean up criteria to protect human health	3	Beyer 1990
The Netherlands	Target value Intervention value	0.3 10	MHSPE 1994
New Jersey	to protect groundwater (not cleanup objectives)	1	New Jersey DEP 1987
Sweden	a guideline value for total Hg in soils	0.5	Lindqvist 1992
Soviet Union	maximum allowable soil concentration	2.1	USSR-SCST 1984
Australia and New Zealand	typical range of background levels; investigation should occur if this level is exceeded	0.001-0.1 (A) 1 (B)	ANZECS 1992
EEC	soil limit value from sewage sludge: pH 6.0-7.0 pH >7.0	1.5 (Agr) 1.5-2.25	EEC 1986

A-background concentrations in soil
B-moderate soil contamination which requires additional study
C-threshold value that requires immediate cleanup

Agr -Agricultural land use
R/P -Residential/Parkland land use
C/I -Commercial/Industrial land use

Table 6. Selected toxicological studies of mercury on soil microbial processes

Species/processes	Effect (a) (% decrease)	Endpoint (exposure period)	Conc mg Hg·kg^{-1}	Form of Hg	pH	Test Substrate	Extraction method	Reference
microorganisms/ CO_2 production	reduction (33%) reduction (16%)	EC (6 h) EC (4 w)	200 0.1	$HgCl_2$	8.1	clay 40%, silt 25%, sand 25%, OM 2.8%	nominal	Landa and Fang 1978
	reduction (41%) reduction (16%)	EC (6 h) EC (4 w)	200 10	$HgCl_2$	6.6	clay 25%, silt 56%, sand 19%, OM 11.5%		
	reduction (37%) reduction (87%)	EC (6 h) EC (4 w)	200 0.1	$HgCl_2$	8.3	clay 25%, silt 48%, sand 27%, OM 2.9%		
	reduction (16%)	EC (6 h)	200	$HgCl_2$	7.5	clay 29%, silt 42%, sand 29%, OM 3.3%		
	reduction (69%)	EC (6 h)	400	$HgCl_2$	8.3	clay 12%, silt 14%, sand 74%, OM 1.6%		
microorganisms/ CO_2 production	reduction reduction (NQ)	NOEC (18 w) EC (18 w)	1.47 14.78	$HgCl_2$	5.9 to 6.3	Clay and sandy soils (clay 7 to 31%, silt 26 to 58%, sand 17 to 67%, OM 1.9 to 20.6%)	nominal	Zelles *et al.* 1986
microorganisms/ O_2 consumption	reduction reduction (28%)	NOEC (67 h) EC (67 h)	70 140	$HgCl_2$	7.37	Sandy loam soil (O.M. 2.86%)	nominal	Tu 1988
N mineralization nitrification N mineralisation nitrification	reduction (26%) reduction (8%) reduction (85%) reduction (21%)	EC (8 years) EC (8 years) EC (8 years) EC (8 years)	50 50 200 200	Hg_2Cl_2	6 ($CaCl_2$)	soil (clay 9%, silt 12%, sand 79% and OC 1.2%)	aqua regia extraction	Wilke 1989
nitrification	reduction (94%) reduction (98%) reduction (97%)	EC (10 d) EC (10 d) EC (10 d)	1003 1003 1003	$HgCl_2$	5.8 7.8 7.4	soil (clay 23 to 34%, silt 39 to 50, sand 16 to 38, OM 4.4 to 9.3)	nominal	Liang and Tabatabai 1978
microorganisms/ CO_2 production	reduction (29%) (33%) reduction (53%) (55%)	EC (2 w) (8 w) EC (2 w) (8 w)	10 10 100 100	$HgSO_4$	4.9	sandy soil (sand 82%, silt 9.9%, clay 5.2%, and OM 3.57%)	nominal	Cornfield 1977

Species/processes	Effect (a) (% decrease)	Endpoint (exposure period)	Conc mg Hg·kg^{-1}	Form of Hg	pH	Test Substrate	Extraction method	Reference
Nitrogen mineralization	reduction reduction (40 to 95%)	NOEC (6 w) EC (6 w)	7.4 74	$HgCl_2$	4.9 to 7.7	Clay soil and dune sand (OM 1.5 to 6.5%)	nominal	Van Faassen 1973
CO_2 production	reduction (20%)	EC (6 w)	74					
Nitrogen mineralization	reduction (57%) reduction reduction (94%)	EC NOEC EC	6 6 60	PMA		Dune sand (OM 1.5%) Clay soils (OM 5.6 to 6.5%)		
Nitrogen mineralization (b)	reduction (73%)	EC (20 d)	1003	$HgCl_2$	5.8	clay 23%, silt 39%, OM 4.4%	nominal	Liang and Tabatabai 1977
	reduction (39%)	EC (20 d)	1003	$HgCl_2$	6.6	clay 45%, silt 54%, OM 5.1%		
	reduction (35%)	EC (20 d)	1003	$HgCl_2$	7.8	clay 30%, silt 54%, OM 6.4%		
	reduction (32%)	EC (20 d)	1003	$HgCl_2$	7.4	clay 34%, silt 50%, OM 9.4%		
CO_2 production	reduction reduction (NQ)	NOEC (48 d) EC (48 d)	14.78 147.8	$HgCl_2$	6.4	Soil (O.M. 3.15%, clay 33.6%)	nominal	Zelles *et al.* 1985

a: The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s)
b: Single concentration study
NQ= Effects not quantifiable

Table 7. Additional toxicological studies of mercury on soil microbial processes

Species/process	Effect (a)	Endpoint	Conc mg Hg·kg^{-1}	Form of Hg	pH	Test Substrate	Extraction method	Reference
enzyme/ H_2 oxidizing activity	inhibition (50%) inhibition (50%)	$P2_{30}$ $P2_{30}$	36 26	$HgCl_2$ $Hg(NO_3)_2$	NA	Fine sandy loam	nominal	Rogers and Pryfogle 1986
CO_2 release	inhibition (NQ)	LOEC	1.3	$HgCl_2$	NA	soil	NA	Wilke 1988
urease activity	inhib. (25 to 29%) inhib. (75 to 98%)	EC EC	100 1003	$HgCl_2$	5.1 to 7.4	soil (clay 17 to 34%, sand 1 to 38% and OM 2.6 to 9.4%)	nominal	Tabatabai 1977
urease activity	inhibition (36%) inhibition (40%) inhibition (71%) inhibition (65%) inhibition (64%) inhibition (77%) inhibition (81%) inhibition (42%) inhibition (38%)	EC EC EC EC EC EC EC EC EC	50 50 30 30 30 120 180 50 50	$HgSO_4$ PMA $HgCl_2$	7.3 6.5 6.5 7 7.3 7.3 7.3 6.5 7.3	soil (clay 20 to 31%, sand 4 to 44%, OM 3.8 to 5.2%)	nominal	Bremner and Douglas 1971
enzyme activity (cellulase, xylanase, invertase etc)	inhibition inhibition (NQ)	NOEC EC	10 to 100 100 to 1000	$HgCl_2$	NA	litter	nominal	Spalding 1979
Arylsulfatase activity	inhib. (34 to 39%) inhibition (94%)	EC EC	501.5 5015	$HgCl_2$	6.2 to 7.6	soil (clay 26 to 34%, silt 34 to 45%, sand 21 to 39%, OM 2.7 to 4.7%)	nominal	Al-Khafaji and Tabatabai 1979
Amidase activity	inhibition (13%) inhib. (27 to46%)	EC EC	100 1003	$HgCl_2$	5.6 to 7.6	clay 28 to 34%, sand 3 to 31%, OM 2.63 to 4.73%)	nominal	Frankenberger and Tabatabai 1981
Phosphatase activity	inhibition inhib. (41 to 63%)	NOEC EC	501.5 5015	$HgCl_2$	5.8 to 7.8	clay 23 to 34%, sand 16 to 38%, silt 39 to 50, OM 4.4 to 9.3%)	nominal	Juma and Tabatabai 1977
CO_2 release	inhibition (25%)	EC	0.06-0.08	$HgCl_2$	NA	soil	NA	Von Stadelmann and Santschi-Fuhrimann 1987

a: The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s)
NQ: Effects not quantifiable
NA: Not available

Table 8. Selected plant and invertebrate toxicological studies for mercury

Species	Effect(a)	Endpoint	Concentration $mg \cdot kg^{-1}$	Form of Hg	Soil pH	Test Substrate	Extraction method	Reference
Radish (*Raphanus sativa*)	Seedling emergence	NOEC LOEC (52% reduction) EC_{25} EC_{50}	51 103 73 103	$HgCl_2$	4 to 4.2	Artificial soil: sand 70 to 75%; clay 16 to 22%;silt 8 to 13%; moisture 80% WHC; OM 4.7 to 10.4%	HNO_3, HCl and H_2O_2	Environment Canada 1995
Lettuce (*Lactuca sativa*)	seedling emergence	NOEC LOEC (27% reduction) EC_{25} EC_{50}	7 12 11 15	$HgCl_2$	4 to 4.2	Artificial soil: ; sand 69 to 75%; clay 16 to 22%;silt 8 to 10%; moisture 80% WHC; OM 4.7 to 6.3%		
Turnip (*Brassica rapa*)	seedling emergence	EC_{50}	50	$HgCl_2$	6.3	sand (clay 3%, OM neglizable)	HCl, HNO_3 and $KMnO_4$	Sheppard *et al.* 1993
	first bloom plant height stem dw	EC_{50} EC_{50} EC_{50}	7 70 40					
	first bloom plant height	EC_{50} EC_{50}	550 850		7.3 7.9	garden soil (clay 46%, OM 15%) clay (clay 43%, OM 4.6%)		
	first bloom, stem dw and no of pods	EC_{50}	1000		7.9	clay (clay 43%, OM 4.6%)		
Lettuce (*Lactuca sativa*)	seedling emergence	EC_{50} EC_{50} EC_{50}	30 500 200		6.3 7.3 7.9	sand (clay 3%, OM neglizable) garden soil (clay 46%, OM 15%) clay (clay 43%, OM 4.6%)		
Earthworm (*Eisenia fetida*)	mortality	NOEC (survival) LOEC (58% mortality) LC_{25} LC_{50}	96 194 130 181	$HgCl_2$	4 to 4.2	Artificial soil: sand 70 to 75%; clay 16 to 22%;silt 8 to 13%; moisture 80% WHC; OM 4.7 to 10.4%	HNO_3, HCl and H_2O_2	Environment Canada 1995
Earthworm (*Lumbricus terrestris*)	mortality	LD_{50} LD_{50}	60 700	$HgCl_2$	6.3 7.3	sand (clay 3%) garden soil(clay 46%, OM 15%)	HCl, HNO_3 and $KMnO_4$	Sheppard *et al.* 1993

Species	Effect(a)	Endpoint	Concentration $mg \cdot kg^{-1}$	Form of Hg	Soil pH	Test Substrate	Extraction method	Reference
Earthworm (*Eisenia fetida*)	mortality mortality(25%) mortality(50%)	NOEC LC LC	100 250 300	$HgCl_2$	7.6	peaty marshland soil and horse manure (1:1)	nominal	Fisher and Koszorus 1992
Bermuda-grass (*Cynodon dactylon*)	plant growth	NOEC LOEC (NQ) LOEC (NQ)	50 8 50	$HgCl_2$	7.6 7.7 4.7	Black clay: O.M. 3.9%, clay 65% Silt loam: O.M. 1.29%, clay 25% Fine sand: O.M. 0.39%, clay 6%	nominal	Weaver *et al.* 1984
Bent grass (*Agrostis canina*)	no toxic effects	NOEC[b]	455	PMA	6.4	garden soil (loam with 3.6% of OM)	HNO_3	Estes *et al.* 1973

a: The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s)
b: The concentration reported here comes from a single-dose study
NQ= Effects not quantifiable

Table 9. Additional plant toxicological studies for mercury

Species	Effect	Endpoint	Conc $mg \cdot kg^{-1}$	Form of Hg	Test Substrate	Reference
Radish (*Raphanus sativa*)	root elongation	NOEC LOEC (25% reduc.) EC_{25} EC_{50}	12 $mg \cdot L^{-1}$ 16 $mg \cdot L^{-1}$ 15 $mg \cdot L^{-1}$ 70 $mg \cdot L^{-1}$	$HgCl_2$	Nutrient solution	Environment Canada 1995
Lettuce (*Lactuca sativa*)	root elongation	NOEC LOEC (50% reduc.) EC_{25} EC_{50}	6 $mg \cdot L^{-1}$ 12 $mg \cdot L^{-1}$ 9 $mg \cdot L^{-1}$ 18 $mg \cdot L^{-1}$			
Pea (*Pisum sativum*)	shoot length, root length, plant growth	NOEC LOEC (NQ)	1 $mg \cdot L^{-1}$ 5 $mg \cdot L^{-1}$	$HgCl_2$	nutrient solution	Beauford *et al.* 1977
Mentha spicata		NOEC LOEC (NQ)	1 $mg \cdot L^{-1}$ 5 $mg \cdot L^{-1}$			
Alfalfa (*Medicago sativa*)	chlorophyll	NOEC LOEC	1 $\mu g \cdot L^{-1}$ 10 $\mu g \cdot L^{-1}$	$HgCl_2$	nutrient solution	Mhatre and Chaphekar 1984
Okra (*Abelmoschus esculentus*)		NOEC LOEC	1 $\mu g \cdot L^{-1}$ 10 $\mu g \cdot L^{-1}$			
Cereal (*Pennisetum typhoideum*)	leaf injury	NOEC LOEC	1 $\mu g \cdot L^{-1}$ 10 $\mu g \cdot L^{-1}$			
	chlorophyll	NOEC LOEC	10 $\mu g \cdot L^{-1}$ 100 $\mu g \cdot L^{-1}$			
Pine (*Picea abies*)	root growth	NOEC LOEC (NQ) LOEC (NQ)	2 $\mu g \cdot L^{-1}$ 200 $\mu g \cdot L^{-1}$ 0.215 $\mu g \cdot L^{-1}$	$HgCl_2$ CH_3HgCl	Nutrient solution	Godbold 1991
Tomato (*Lycopersicom esculentum*)	plant growth	LOEC(NQ)	0.01 $mg \cdot L^{-1}$	CH_3HgOH	modified Hoagland's solution	Haney and Lipsey 1973
Beans (*Phaseolus vulgaris*)		LOEC	0.05 $mg \cdot L^{-1}$			
15 test species	seed germination and seedling growth	NOEC	14 $\mu g \cdot L^{-1}$	HgO vapour	Air saturation of 14 $\mu g \cdot L$-1	Siegel *et al.* 1984

NQ= Effects not quantifiable

Table 10. Additional invertebrate toxicological studies for mercury

Species	Effect (a)	Endpoint	Conc ($mg \cdot kg^{-1}$)	Form of Hg	Soil pH	Test Substrate	Extraction Method	Reference
Earthworm (*Octochaetus pattoni*)	mortality	LC_{50} - 5d LC_{50} - 10d LC_{50} - 15d LC_{50} - 30d LC_{50} - 40d LC_{50} - 50d LC_{50} - 60d	>5 2.39 1.66 1.51 1.04 0.91 0.79	$HgCl_2$	NA	soil mixed with animal dung	nominal	Abbasi and Soni 1983
Red wiggler (*Eisenia fetida savigny*)	mortality mortality (21%)	NOEC EC	3.25 9.5	CH_3HgCl	NA	potting soil in petri dishes	nominal	Beyer *et al.* 1985
Slug (*Arion ater*)	food consumption	NOEC	10 $mg \cdot kg^{-1}$	$HgCl_2$	--	0-1000 mg kg^{-1} in diet	--	Marigomez *et al.* 1986
Aphids (*Macrosiphum gei Ashmead*) /reproduction	(31% reduction) (41% reduction) (100% reduction)	EC_{31} EC_{41} EC_{100}	0.004 $mg \cdot L^{-1}$ 0.02 $mg \cdot L^{-1}$ 0.04 $mg \cdot L^{-1}$	CH_3HgOH (MMH)	--	aphids fed on tomato seedlings grown in a culture solution with MMH	--	Haney and Lipsey 1973

a: The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s)
NA: not available

Table 11. Selected mammalian and avian toxicological studies for mercury

Species	Effect(a)	Endpoint	Diet concentration (mg Hg.kg^{-1})	Average dose (mg/kg bw . day^{-1})	Form of Hg	Exposure period	Reference
Pig (16 to 28 kg each)	CNS depression, weight loss, fever, anorexia, and death death (75%)	NOEL LOEL	1.7 13.4	(-) (-)	MMD	single dose	Piper *et al.* 1971
Pheasant	mortality	LD_{50}	65	(-)	PMA	single dose	Mullins *et al.* 1977
Rat (46 to 50 g weight)	growth reduction, kidney damage, histochemical changes and mortality growth reduction, kidney damage, histochemical changes and mortality (male 37% and female 48%)	NOEL LOEL	0.08 to 0.4 2	0.016 to 0.079 0.393	MeHgCl	2 year	Verschuuren *et al.* 1976b
Rat (31 to 41 g weight)	toxic signs, weight loss and food intake reduction	NOEL LOEL (NQ)	10 20	1.96 3.93	MeHgCl	2 and 12 weeks	Verschuuren *et al.* 1976a
Pig	clinical signs, severe lesions and death	NOEL LOEL (NQ)	(-) (-)	0.19 0.38	MMD and EMC	60 to 90 days	Tryphonas and Nielsen 1973
Coturnix (*Coturnix japonica*)	mortality	LD_{50} (acute) LC_{50} (subacute) LD_{50} (acute) LC_{50} (subacute)	14.4 37.6 31 3764	(-) 17.5 (-) 1751	MeHgCl $HgCl_2$	single dose 5 days single dose 5 days	Hill and Soares 1984
Mallard	egg laying, hatching, and mortality of hatchlings	NOEC LC_{50}	0.5 3	0.1 0.6	MMD	21 weeks	Heinz 1974
Bobwhite chick	mortality (80%)	LC	16	2.88	CH_3HgCl	6 weeks	Spann *et al.* 1986
Zebra finches	mortality mortality (25%)	NOEC LOEL	(-) (-)	0.88 1.75	CH_3Hg	76 days	Scheuhammer 1988
Peking duck (*Anas Platyrhynchos*)	reduction in wt	NOEL LOEL(NQ)	0.48 3.76	0.058 0.45	MMC	7 weeks	Bhatnagar *et al.* 1982

MMC=Methyl mercury chloride, MMD=Methylmercuric dycyandiamide, EMC=Ethylmercuric chloride
a: The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s)
NQ= Effects not quantifiable

Table 12. Additional acute toxicological studies of mercury on terrestrial birds

Species	Effect	Endpoint	Dose (mg Hg·kg^{-1} bw)	Form of Hg	Exposure period (1)	Exposure route	Reference
Japanese Quail	mortality	LC_{50} LD_{50}	614 169	PMA[2]	5 days acute	oral	Hill and Camardese 1986
Mallard Duck	mortality	LD_{50} LD_{50}	53 878	MeHg[3] PMA[2]	acute acute	oral oral	Hudson *et al.* 1984

1 acute as single oral dose
2 Phenylmercuric acetate
3 Panogen

Table 13. Additional subacute toxicological studies of mercury on terrestrial birds

Species	Effect (a)	Endpoint	Concentration	Form of Hg	Exposure Period	Exposure Route	Reference
Japanese Quail	enzyme activity and mortality	EC_{50}(ASAT sublethal) EC_{50}(ASAT sublethal) EC_{50}(OCT sublethal) EC_{50}(OCT sublethal) EC_{50}(LDH sublethal) EC_{50}(LDH sublethal)	4 6.7 3.2 47 0.8 2.2	CH_3HgCl $HgCl_2$ CH_3HgCl $HgCl_2$ CH_3HgCl $HgCl_2$	9 weeks	oral	Hill and Soares 1984
Red-Tailed Hawks	neurotoxicity, weakness and behavioral changes	LOEC (NQ)	10 $mg \cdot kg^{-1}$	MMD[1]	4-12 weeks	oral	Fimreite and Karstad 1971
Mallard Ducks	growth and survival of hatchlings	NOEC	0.5 $mg \cdot kg^{-1}$	MMD	2 years	oral	Heinz 1976
Japanese Quail	mortality	NOEC	32 $mg \cdot kg^{-1}$	$HgCl_2$	1 year	oral	Hill and Shaffner 1976
Pheasants	hatching, growth and survival of chicks	LOEC (NQ)	20 mg $Hg \cdot kg^{-1}$	PMA[2]	--	oral	Mullins *et al.* 1977
Chicken eggs	toxicity level toxicity level	TL_{50}[3] TL_{50}[3]	0.1-0.5 $mg \cdot kg^{-1}$ 1.0 $mg \cdot kg^{-1}$	CH_3Hg $HgCl_2$	--	yolk injection	Birge and Roberts 1976
Chickens	growth and hatching numbers	LOEC(NQ)	10 $mg \cdot kg^{-1}$	CH_3HgCl	3 weeks	oral	Scott 1977
Black Ducks	hatching and duckling mortality (75%)	LD	3 $mg \cdot kg^{-1}$	CH_3Hg	28 weeks	oral	Finley and Stendell 1978
Mallard eggs	embryonic growth	reduced growth	50 $mg \cdot kg^{-1}$	CH_3HgCl	10 days	surface application to egg	Hoffman and Eastin 1981
Peking Ducks	liver damage and neurotoxicity	NOEC EC (NQ)	3.76 $mg \cdot kg^{-1}$ 13.43 $mg \cdot kg^{-1}$	CH_3HgCl	12 weeks	oral	Bhatnagar et al. 1982
Chicken eggs	hatching and mortality	LD_{50}	40-50 $mg \cdot kg^{-1}$	CH_3HgCl	--	yolk injection	Greener and Kochen 1983

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

1. MMD=methyl mercuric dicyandiamide
2. PMA = Phenyl mercuric acetate
3. TL_{50} = 50 percent survival toxicant level

ASAT=aspartate aminotransferase, OCT=ornithine carbamoyltransferase, LDH=lactate dehydrogenase,
NQ= Effects not quantifiable

Table 14. Additional mammalian toxicological studies for mercury

Species	Effect	Endpoint	Concentration	Form of Hg	Exposure Period	Exposure Route	Reference
Cat	neurological impairment	NOEL LOEL (NQ)	20 $\mu g \cdot kg^{-1}$ b.w. 46 $\mu g \cdot kg^{-1}$ b.w.	CH_3HgCl	2 years	oral	Charbonneau *et al.* 1976
Mouse	chronic nephropathy	NOEL LOEL (NQ)	0.4 $mg \cdot kg^{-1}$ feed 2 $mg \cdot kg^{-1}$ feed	CH_3HgCl	2 years	oral	Mitsumori *et al.* 1990
Mice	kidney damage and lipid peroxidase, 50% reduction	NOED LOED	50 $\mu mol \cdot kg^{-1}$ 200 $\mu mol \cdot kg^{-1}$	$HgCl_2$	single exposure	oral	Nielsen *et al.* 1991
Rhesus Monkey	liver and kidney functions	NOEL	125 $\mu g \cdot kg^{-1}$ b.w.	CH_3HgO	1 year	oral	Chen *et al.* 1983

NQ= Effects not quantifiable

Table 15. Selected data on the accumulation of mercury in plants used for the BCF calculation

Species	Hg Compound	Tissue Type	Bioconcentration factor	References
Broccoli *(Brassica oleracea)*	$HgCl_2$	leaves roots	0.002 0.09	John 1972
Pea *(Pisum Sativa)*	$HgCl_2$	roots	0.07	John 1972
Cauliflower *(Brassica oleracea)*	$HgCl_2$	leaves roots	0.003 0.12	John 1972
Spinach *(Spinacia oleracea)*	$HgCl_2$	leaves roots	0.035 0.05	John 1972
Lettuce *(Lactuca sativa L.)*	$HgCl_2$	roots leaves	0.019 0.0023	John 1972
Oat *(Avena sativa L.)*	$HgCl_2$	leaves	0.00995	John 1972
Radish *(Raphanus sativus L.)*	$HgCl_2$	tops	0.0293	John 1972
Carrot *(Daucus carota L.)*	$HgCl_2$	tops roots	0.0036 0.0529	John 1972
Rutabagas *(Brassica napobrassica)*	PMA and EMC	tops roots	0.50 0.013	Gracey and Stewart 1974
Alfalfa *(Medicago sativa)*	PMA	main roots	0.245	Gracey and Stewart 1974
Alfalfa *(Medicago sativa)*	Mercuric acetate	main roots	0.704	Gracey and Stewart 1974
Alfalfa *(Medicago sativa)*	$HgSO_4$	main roots	0.599	Gracey and Stewart 1974
Alfalfa *(Medicago sativa)*	$HgCl_2$	main roots	0.501	Gracey and Stewart 1974
Alfalfa *(Medicago sativa)*	5% Hg Fungicide (PMA & EMC)	main roots	0.345	Gracey and Stewart 1974
Bermudagrass *(Cynodon dactylon L.)*	$HgCl_2$	roots roots roots	5 1.6 16	Weaver *et al.* 1984
Chloris barbata Sw.	Hg	root shoot root shoot root shoot root shoot	0.44 0.10 0.19 0.05 0.11 0.02 0.1 0.02	Lenka *et al.* 1992
Argemone mexicana L.	Hg	root shoot	0.03 0.02	Lenka *et al.* 1992
Cynodon dactylon L.	Hg	root shoot root shoot root shoot root shoot	0.55 0.22 0.40 0.09 0.29 0.09 0.32 0.06	Lenka *et al.* 1992

Species	Hg Compound	Tissue Type	Bioconcentration factor	References
Broccoli *(Brassica oleracea)*	$HgCl_2$	leaves	0.002	John 1972
		roots	0.09	
Cyperus rotundus L.	Hg	root	1.15	Lenka *et al.* 1992
		shoot	0.33	
		root	0.42	
		shoot	0.11	
		root	0.36	
		shoot	0.06	
Croton bonplandianum Baill.	Hg	root	0.03	Lenka *et al.* 1992
		shoot	0.05	
		root	0.11	
		shoot	0.2	
		root	0.04	
		shoot	0.07	
		root	0.04	
		shoot	0.04	
		root	0.004	
		shoot	0.003	
Evolvulus alsinoides L.	Hg	root	0.32	Lenka *et al.* 1992
		shoot	0.22	
		root	0.46	
		shoot	0.39	
Justicia simplex D.	Hg	root	0.18	Lenka *et al.* 1992
		shoot	0.2	
		root	0.24	
		shoot	0.24	
Tribulus terrestris L.	Hg	root	0.22	Lenka *et al.* 1992
		shoot	0.36	

Table 16. Environmental Guidelines for Inorganic Mercury

Guidelines		Land Use		
		Agriculture mg Hg·kg^{-1}	Residential/Parkland mg Hg·kg^{-1}	Commercial/Industrial mg Hg·kg^{-1}
SQG_{SC}	20th percentile (a)	4	4	Not applicable
	25th percentile (a)	10	10	30
	30th percentile	12.5	12.5	50
	35th percentile	Not applicable	57	73
Nutrient and energy cycle check		15	15	57
SQG_I		19 (b)	Not applicable	Not applicable
SQG_E		10	10	30
CCME interim criteria (CCME 1991)		0.8	2	10

(a): According to the CCME 1996 protocol the SQG_{SC} for Agriculture and Residential/Parkland corresponds to the 25th percentile of the effects and no effects distribution using the Weight of Evidence method, while the SQG_{SC} for Commercial/Industrial land use corresponds to the 25th percentile of the effects distribution. The other percentiles were provided for comparison purpose only.

(b): There are insufficient data to calculate the soil and food ingestion check for inorganic mercury. Using data available for organic mercury, a food and soil ingestion check of 19 mg Hg/kg soil has been calculated.

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