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**Canadian Soil Quality Guidelines for**

**Lead: 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 lead. 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 lead and a review of the sources and emissions in Canada. Chapter 3 discusses lead's distribution and behavior in the environment while chapter 4 reports the toxicological effects of lead on microbial processes, plants, and animals. These informations are used in chapter 5 to derive soil quality guidelines for lead 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 lead, the environmental soil quality guideline ( $SQG_E$ ) relative to agricultural land use is  $70 \text{ mg}\cdot\text{kg}^{-1}$  soil, it is  $250 \text{ mg}\cdot\text{kg}^{-1}$  soil for residential/parkland land use and  $400 \text{ mg}\cdot\text{kg}^{-1}$  soil for commercial and industrial land uses. These environmental soil quality guidelines are optimized for soils within the pH range of 4 to 8.8 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 plomb 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 plomb 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 plomb, la recommandation pour la qualité des sols en vue de la protection de l'environnement (RQS<sub>E</sub>) relative aux terrains à vocation agricole est de 70 mg·kg<sup>-1</sup> de sol, elle est de 250 mg·kg<sup>-1</sup> de sol pour les terrains à vocation résidentielle/parc et elle est de 400 mg·kg<sup>-1</sup> 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 8.8 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 lead, its distribution and behaviour in the environment, and its toxicological effects on plants, microbial processes and animals. This information is used to derive guidelines for lead to protect ecological receptors according to the processes outlined in 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

Lead (Pb) is a metal of the subgroup IVB in the periodic table. It has an atomic number of 82, an atomic weight of 207.19, and a valence of +2 or +4. Lead is a lustrous, silvery metal that tarnishes in the presence of air to become a dull, bluish grey. The metal has a relatively low melting point of 327.5°C and a boiling point of 1740°C (Weast 1977). It derives its chemical symbol, Pb, from the Latin word "plumbum", meaning "waterworks", because of its common use in pipes during Roman times (Fleischman 1993). The solubility of metallic lead is very low, while the solubilities of other lead compounds range from very soluble  $[\text{Pb}(\text{NO}_3)_2]$  to extremely insoluble  $[\text{Pb}_3(\text{PO}_4)_3\text{Cl}]$  (Lindsay 1979).

Lead is common in the earth's crust, which contains about  $15 \text{ mg}\cdot\text{kg}^{-1}$  of this metal (Fleischman 1993). Soils have an average content of about  $10 \text{ mg}\cdot\text{kg}^{-1}$  (Lindsay 1979). In solution,  $\text{Pb}^{2+}$  is the most common species, although at high pH ( $>8$ )  $\text{PbOH}^+$  becomes more soluble. Similarly, high concentrations of halides ( $\text{Cl}^-$ ,  $\text{Br}^-$ ,  $\text{F}^-$ ,  $\text{I}^-$ ) or sulphate ( $\text{SO}_4^{2-}$ ) will render the corresponding lead compounds more soluble than  $\text{Pb}^{2+}$ . In soils, Pb will tend to precipitate in the form of lead carbonate.

$[\text{Pb}_3(\text{CO}_3)_2\cdot(\text{OH})_2]$  because of the elevated  $\text{CO}_2$  partial pressures generated by organic matter



decomposition and root respiration. In the presence of phosphate ( $\text{PO}_4$ ), even more insoluble forms will precipitate [i.e.,  $\text{Pb}_5(\text{PO}_4)_3\text{Cl}$ ] (Lindsay 1979). The physical and chemical properties of lead and its principal compounds are listed in Table 1.

## 2.2 Analytical Methods

Inductively coupled plasma-atomic emission spectroscopy (U.S. EPA Method 6010) is the recommended analytical method for the measurement of lead in soils (CCME 1993). The detection limit and precision of this method are  $42 \mu\text{g}\cdot\text{L}^{-1}$  and 18%, respectively. There are three analytical methods for the measurement of lead in water, wastewater, and soil extracts: direct air-acetylene flame method (U.S. EPA Method 3111B), electrothermal atomic absorption spectrometric method (U.S. EPA Method 3113B), and inductively coupled plasma method (U.S. EPA Method 3120B). The detection limits are, respectively,  $50 \mu\text{g}\cdot\text{L}^{-1}$ , not available, and  $40 \mu\text{g}\cdot\text{L}^{-1}$ . (For further details, refer to CCME [1993]). Following corrections for a 1 g soil sample digestion with the extract being diluted to 100 mL, detection limits in soils of 5 and  $4 \text{ mg}\cdot\text{kg}^{-1}$ , respectively, would be calculated.

## 2.3 Production, Uses, and Global Sources

### *Production and Canadian Exports and Imports*

The most important primary lead ore is galena ( $\text{PbS}$ ). This sulfide ore is found in hydrothermal veins in dolomites, limestones, and other sedimentary rocks. Primary ores account for over 95% of mined lead production, and  $\text{PbS}$  is the predominant ore recovered (EPS 1985). Other common primary ores are cerussite ( $\text{PbCO}_3$ ) and anglesite ( $\text{PbSO}_4$ ). Lead production from ore is often associated with zinc production as lead and zinc are frequently found together in nature.

There is a downward trend in consumption of lead in the western world. From 1991 to 1992, a decrease of 0.8% was observed (Keating 1992). The International Lead and Zinc Study Group found that Canada, along with Mexico and Sweden, were increasing outputs of primary lead production. This increase offsets reductions in production by the United States and Yugoslavia. In Canada, new mines are opening in British Columbia, Yukon, and the Northwest Territories. In addition, existing smelters are returning to their normal output capacities. Therefore, Canadian lead production is expected to continue its increase (Keating 1992).

World refined lead production was reported as 5 588 000 t in 1991, of which Canada was responsible for 276 000 t (5%). In 1992, lead mine output (primary production) for Canada was 342 000 t, up from 277 000 t in 1991. In terms of refined production, there was an output of 146 000 t of primary-refined lead metal (up 40 000 t from 1991) and an output of 104 000 t of secondary-refined lead metal from recycled materials (down 2000 t from 1991). Lead is one of the most recycled metals in the world (Energy, Mines and Resources Canada 1992a).

In 1991, 233 000 t of lead were exported from Canada. Japan imported 20%, Italy 19%, and the

United States 16% of Canadian lead ores and concentrates. The other main importing countries of Canadian lead products were India, South Korea, Germany, and Australia (Energy, Mines and Resources Canada 1992b). Canada imported 47 000 t of lead in 1992, primarily in the form of lead waste and scrap (82%).

### *Uses*

Total (primary and secondary) lead consumption in Canadian industry in 1991 was reported to be 68 000 t, compared to 71 000 t in 1990. This consumption was due to the production of antimonial lead, batteries and battery oxides, lead for chemical uses (white and red tetraethyl and litharge), copper alloys (brass, bronze, etc.), lead alloys (solders, babbitts, type metal), and semifinished products, such as pipe, sheet, traps, bends, and blocks for caulking and ammunition (Energy, Mines and Resources Canada 1992b).

Lead and its various compounds are used in a variety of applications. For example, basic lead chromate ( $\text{PbO} \cdot \text{PbCrO}_4$ ) and lead chromate ( $\text{PbCrO}_4$ ) are used in the production of pigments. Lead silicate ( $\text{PbO} \cdot \text{SiO}_2$ ) is used in glass and ceramic production. Lead sulfate ( $\text{PbSO}_4$ ) is used in the production of pigments, batteries, and in lithographic processes. Tetraethyl lead [ $\text{Pb}(\text{C}_2\text{H}_5)_4$ ] was widely used as a gasoline anti-knock additive until regulations banned the use of lead in gasoline. Tetramethyl lead [ $\text{Pb}(\text{CH}_3)_4$ ] is used to fill Geiger counters (Jaques 1985). Lead arsenate ( $\text{PbAsO}_4$ ) was used as an insecticide, especially in fruit tree orchards, from 1910 until 1975, but is no longer marketed (Jaques 1985).

Lead is also associated with phosphate fertilizers such as diammonium phosphate and fluid fertilizer, which contain North Carolina phosphate rock with lead concentrations of  $4.7\text{--}5.2 \text{ mg} \cdot \text{kg}^{-1}$ . Triple superphosphate has a lead concentration of  $238 \text{ mg} \cdot \text{kg}^{-1}$ , and N-P-K mixture has a lead concentration of  $444 \text{ mg} \cdot \text{kg}^{-1}$ . The addition of diammonium phosphate to soil at a rate of  $500 \text{ kg} \cdot \text{ha}^{-1}$  per year for 100 years would add a total of 0.24 kg of lead per hectare (Mortvedt and Giordano 1977).

### *Global Sources of Lead*

The concentration of lead in the earth's crust is approximately  $15 \text{ mg} \cdot \text{kg}^{-1}$  (Fleischman 1993). There are four stable isotopes of lead that occur in nature: Pb-204 (1.48%), Pb-206 (23.6%), Pb-207 (22.6%), and Pb-208 (52.3%) (NRCC 1973). Pb-206, Pb-207, and Pb-208 are formed as the end products of uranium, actinium, and thorium decay, respectively (Morris 1992).

Lead is found at various concentrations in most soil-forming rocks (Adriano 1986). Basaltic igneous rocks contain lead at concentrations of  $2\text{--}18 \text{ mg} \cdot \text{kg}^{-1}$ ; granitic igneous rocks,  $6\text{--}30 \text{ mg} \cdot \text{kg}^{-1}$ ; shales and clays,  $16\text{--}50 \text{ mg} \cdot \text{kg}^{-1}$ ; black shales,  $7\text{--}150 \text{ mg} \cdot \text{kg}^{-1}$ ; and sandstones,  $<1\text{--}31 \text{ mg} \cdot \text{kg}^{-1}$ . It is therefore clear that most soils will have natural background levels of lead. However, ultramafic rocks, deep-sea clays, and limestones are not reported to contain lead.

Nriagu (1989) reported atmospheric emissions of trace metals from natural sources. Total emissions range from 970 to 23 000 t per year, with windborne soil particles being the primary contributor at

about 30% of the total. Volcanoes are responsible for approximately 25% of total natural emissions, while forest fires and sea salt spray account for 15% and 10%, respectively. Biogenic sources such as nonmethane natural hydrocarbons (NMHCs), continental particulates, and continental volatiles were estimated to contribute 14% of the total.

Commercial fertilizers, pesticides, land application of sewage sludge, animal wastes from animal production, coal residues, municipal refuse incineration, wastewaters, mining and smelting, and auto emissions all contribute to the earth's lead burden (Nriagu and Pacyna 1989). Approximately 479 000–1 113 000 t of lead per year are released to the soil worldwide (Nriagu and Pacyna 1989). Approximately 36% of this is the result of "wastage of commercial products", which is described as discarded metals on land. Another 30% originates from atmospheric fallout. Other sources of lead to soil, in decreasing order of importance, are the disposal of coal fly ash and bottom ash, urban refuse, logging and other wood wastes, solid wastes (metal manufacturing), animal wastes, municipal sewage sludge, agricultural and food wastes, peat (agricultural and fuel uses), and fertilizer (Adriano 1986). Distance from the emitting source will influence regional soil lead concentration levels (Dumontet et al. 1989). Natural sources account for only a small portion of total lead deposited to soils.

## **2.4 Levels in the Canadian Environment**

### **2.4.1 Anthropogenic Inputs into the Canadian Environment**

A detailed overview of the distribution of lead in the Canadian environment can be obtained from two documents released by the National Research Council of Canada in the 1970s (NRCC 1973; NRCC 1978). More recent information on the sources and releases of lead to the Canadian environment was reported in a 1982 national inventory of lead emissions, which estimated the total annual release at 73 528 t (Jaques 1985). Of that total, 83.6% (61 726 t) was from solid waste, 15.6% (11 466 t) from anthropogenic gaseous emissions, and 0.76% (562 t) from effluents released to surface waters. A summary of anthropogenic lead emissions is presented in Table 2.

Lead emitted to the atmosphere or released into streams and surface waters is considered more of a threat to the environment than lead in solid wastes since the former is more likely to come into contact with biota and is usually more bioavailable than the latter. In 1982, 63.1% of lead released to the atmosphere was from gasoline combustion and production, 14.9% from copper and nickel production, 8.4% from mining and milling, 5.2% from the iron and steel, 2.7% from lead and zinc production, 1.9% from ferrous foundries, and 3.8% from miscellaneous industries (Jaques 1985).

### **2.4.2 Distribution of Lead in the Canadian Environment**

Evaluation of the levels of both background and anthropogenic lead in air, soil, water, sediment, and biota provides a means of determining the routes and magnitudes of exposures to environmental receptors. These data, in conjunction with detailed toxicological information, can be used to assess

the hazards associated with exposure to lead for terrestrial and aquatic organisms in the Canadian environment.

### *Air*

The National Air Pollution Surveillance network reported that from 1975 to 1983 the annual Canadian geometric mean air lead concentration fell from  $0.55 \mu\text{g}\cdot\text{m}^{-3}$  to  $0.25 \mu\text{g}\cdot\text{m}^{-3}$  and the corresponding composite 24-h maxima fell from  $2.5 \mu\text{g}\cdot\text{m}^{-3}$  to  $1.1 \mu\text{g}\cdot\text{m}^{-3}$  (EPS 1985). The ambient air lead particulate concentration is closely linked to the use of lead in gasolines. From 1975 to 1983, the annual geometric mean lead concentration in air decreased linearly with the decrease in the use of lead additives in gasoline (EPS 1985).

### *Soil*

Background levels for total lead in Canadian soils have been estimated by different researchers and reflect those measured in uncontaminated soils remote from ore bodies. Nriagu (1978) suggested a mean of  $12 \text{ mg}\cdot\text{kg}^{-1}$  for Canadian soils, while McKeague and Wolynetz (1980) reported a mean of  $20 \text{ mg}\cdot\text{kg}^{-1}$ . These background levels are similar to those reported for the United States ( $18\text{--}20 \text{ mg}\cdot\text{kg}^{-1}$ ) and world soils ( $10\text{--}25 \text{ mg}\cdot\text{kg}^{-1}$ ) (Norrish 1975; Aubert and Pinta 1977). However, McKeague and Wolynetz (1980) found significant differences in the mean background levels of soils sampled from different regions within Canada. Higher values were reported for the St. Lawrence Lowlands ( $25 \text{ mg}\cdot\text{kg}^{-1}$ ) and the Appalachian and Canadian Shield regions ( $21 \text{ mg}\cdot\text{kg}^{-1}$ ), while lower levels were found in the Interior Plains ( $15 \text{ mg}\cdot\text{kg}^{-1}$ ) and Cordilleran ( $16 \text{ mg}\cdot\text{kg}^{-1}$ ) areas (McKeague and Wolynetz 1980).

The amount of total lead in agricultural soils depends on the parent material and anthropogenic input to the soil. Warren et al. (1970) suggested that normal Canadian agricultural soils range between  $<1$  and  $12 \text{ mg}\cdot\text{kg}^{-1}$ . A study of 296 agricultural soils in Ontario produced a mean of  $46 \text{ mg}\cdot\text{kg}^{-1}$ , with values ranging from  $1.5$  to  $888 \text{ mg}\cdot\text{kg}^{-1}$  (Frank et al. 1976). Soils in fruit orchards had the highest lead concentrations (mean of  $123 \text{ mg}\cdot\text{kg}^{-1}$ ) resulting from the use of lead arsenate pesticides, while other cropped soils averaged  $14 \text{ mg}\cdot\text{kg}^{-1}$ . Similarly, levels of lead in orchard soils in Nova Scotia ( $>50 \text{ mg}\cdot\text{kg}^{-1}$ ) were higher than those in nonorchard samples (Chisholm and Bishop 1967). Lead levels in agricultural soils close to settlements in Alberta range from  $2$  to  $28 \text{ mg}\cdot\text{kg}^{-1}$ , with a mean of  $9 \text{ mg}\cdot\text{kg}^{-1}$  (J. Lutwick 1993, Alberta Environmental Protection, pers. com.).

Ecosystems often become contaminated in areas where ore smelters are located. Hogan and Wooton (1984) found high levels of lead in the soil of a boreal forest less than  $10 \text{ km}$  from a copper-zinc smelter in Manitoba. Soil lead levels were elevated up to  $35 \text{ km}$  from the emission stack. Bisessar (1982) reported lead concentrations of up to  $28\,000 \text{ mg}\cdot\text{kg}^{-1}$  in soils around a secondary lead smelter, with values of  $703 \text{ mg}\cdot\text{kg}^{-1}$  in soils located  $1 \text{ km}$  south of the smelter.

In uncontaminated Canadian mineral soils such as podzols, gleysols, and luvisols, total levels of lead generally increase down the profile of the soil (Dudas and Pawluck 1980; McKeague and Wolynetz 1980). However, organic soils in the major soil groups can have up to three times as much lead as

nonorganic soils (Nriagu 1978), with the amount of lead in surface horizons increasing with the content of organic matter (McKeague and Wolynetz 1980). Soon and Abboud (1990) compared 52 surface soils and 11 subsoils in cultivated agricultural areas of northwestern Alberta and reported that the mean total lead content of the surface soils ( $13.4 \text{ mg}\cdot\text{kg}^{-1}$ ) was lower than the subsoils ( $16.6 \text{ mg}\cdot\text{kg}^{-1}$ ). However, when surface soils were divided into groups with high or low organic content, the more organic soils (black and dark grey soils and gleysols) had higher total lead concentrations (mean of  $15.4 \text{ mg}\cdot\text{kg}^{-1}$ ) than the less organic soils (grey luvisols and solonetz soils; mean of  $11.6 \text{ mg}\cdot\text{kg}^{-1}$ ). In a study of farmed organic soils (>85% organic matter) from the Holland Marsh area near Toronto, the surface soils contained  $22 \text{ mg}\cdot\text{kg}^{-1}$  compared with  $10 \text{ mg}\cdot\text{kg}^{-1}$  at a depth of 48 cm (Czuba and Hutchinson 1980).

Dudas and Pawluk (1980) looked at the amounts of lead in different fractions of uncontaminated chernozem and luvisol soils from Alberta. Concentrations of lead were higher in the clay ( $22\text{--}30 \text{ mg}\cdot\text{kg}^{-1}$ ) than the silt ( $15\text{--}18 \text{ mg}\cdot\text{kg}^{-1}$ ) and sand ( $7\text{--}12 \text{ mg}\cdot\text{kg}^{-1}$ ) fractions. Soon and Abboud (1990) found that the amounts of total and bioavailable (DTPA-extractable) lead were positively correlated with the clay fraction of cultivated agricultural soils. Bioavailability of lead also increased with decreasing pH.

### *Sediments*

The concentration of lead in sediments varies in regions across Canada. For example, lead concentrations range from 60 to  $154 \text{ mg}\cdot\text{kg}^{-1}$  in sediments within the depositional zone of the Great Lakes (IJC 1977), from 9 to  $314 \text{ mg}\cdot\text{kg}^{-1}$  for sediments in the Ottawa River (Oliver and Agemian 1974), and from 4 to  $63 \text{ mg}\cdot\text{kg}^{-1}$  for open space and residential sediments of mountain streams in Burnaby, British Columbia.

### *Water*

Concentrations of lead in Canadian surface waters range from 1 to  $77 \mu\text{g}\cdot\text{L}^{-1}$  (NAQUADAT 1985). The average lead concentration in Canadian tap water is approximately  $7.6 \mu\text{g}\cdot\text{L}^{-1}$ , with 98% of 266 sampling stations having concentrations under  $50 \mu\text{g}\cdot\text{L}^{-1}$  (NAQUADAT 1976).

## **2.5 Existing Criteria and Guidelines**

Existing guidelines, criteria, or standards for lead in soil from provincial, national, and international agencies are summarized in Table 3.

## **3. ENVIRONMENTAL FATE AND BEHAVIOUR**

### **3.1 Soil**

The greatest risk of lead exposure to wild animals is through ingestion of lead particles from soil or

dust. The lead in contaminated soils originates from (1) the combustion of leaded gasoline and the subsequent deposition of lead in the forms of  $\text{Pb}^0$ ,  $\text{PbSO}_4$ , and other compounds; and (2) industries involved in lead refining, manufacturing, or recycling, in which case lead may be present as lead oxides ( $\text{PbO}$ ,  $\text{Pb}_3\text{O}_4$ ) or lead sulfates (Biggins and Harrison 1980).

Lead can pose a threat to the environment if it moves through the soil and contaminates groundwater and surface waters, or if it is transferred to biota. Many factors influence the mobility and bioavailability of lead: pH, soil texture (especially clay content), and organic matter content. Since dissolved lead in soils is commonly in the form of  $\text{Pb}^{2+}$ , the adsorption on cation exchange sites of clays or organic matter can decrease the mobility and availability of lead in the short term. Erosion of soils by wind or water is an important pathway by which lead-contaminated soils can migrate and contaminate the surrounding environment.

As discussed previously, lead is added to soils as a result of fallout from the atmosphere by either accidental or deliberate dumping of lead-containing wastes, or from the addition of pesticides and fertilizers that contain lead. The form of the metal that is being added to the soil will affect its solubility and initial mobility. For example, lead chlorides, lead acetates, and lead nitrates are readily soluble in the soil environment and will be leached into soil. Lead oxides, although less soluble than the salts, are still more soluble than some of the lead compounds that form in soils. Metallic lead is relatively insoluble but can be oxidized to form  $\text{PbO}$  in soils, which can then be dissolved or transformed into a more stable compound. In aerobic soils, weathering of these very soluble lead compounds results in the formation of more stable compounds such as  $\text{Pb}_3\text{CO}_3(\text{OH})_2$  (Lindsay 1979). In anaerobic soils, the reduction of  $\text{SO}_4^{2-}$  to  $\text{S}^{2-}$  leads to the formation of lead sulfide ( $\text{PbS}$ ), a very insoluble, nonreactive lead species. Even in composted municipal solid waste, the solubility of lead decreases as the material ages and becomes more stabilized (Leita and De Nobili 1991).

In soils, lead solubility seems to be controlled by relatively insoluble compounds such as  $\text{PbCO}_3$ ,  $\text{Pb}(\text{OH})_2$ ,  $\text{Pb}_3(\text{PO}_4)_2$ , or  $\text{Pb}_5(\text{PO}_4)_3\text{OH}$ , which have a pH-dependent solubility in contrast to the lead salts normally used in toxicity testing, whose solubilities are not dependent on pH (Santillan-Medrano and Jurinak 1975). For example,  $\text{Pb}(\text{OH})_2$  would maintain a concentration of  $\text{Pb}^{2+}$  of about  $300 \text{ g}\cdot\text{L}^{-1}$  in solution at pH 4.0, but at pH 7.0, the concentration would be only  $0.0003 \text{ g}\cdot\text{L}^{-1}$ . If, in fact, the dissolved lead is responsible for toxicity to plants and animals, the pH of the soil environment must play a crucial role.

There seems to be no conclusive evidence as to which forms of lead are available to soil biota. The evidence relating the effects of soil properties, such as pH, cation exchange capacity, and the presence of ligands capable of binding lead (e.g.,  $\text{PO}_4$  or organic matter), all suggests that the amount of free metal (i.e.,  $\text{Pb}^{2+}$ ) in solution is the critical fraction (Adriano 1986; Kabata-Pendias and Pendias 1992). Hardiman et al. (1984) found that the amount of lead in a soil solution was 5 times the amount absorbed by beans from the transpiration stream. The authors concluded that only a fraction of the dissolved lead was available and the rest strongly bound in organic or inorganic macromolecules. Numerous studies have shown, however, that good correlations exist between uptake of metal by plants and animals and the amounts extracted from different "pools". Xian (1989) grew cabbage on contaminated soils and found a correlation between the amount of lead absorbed and the amount of

exchangeable carbonate in the soil. Davies (1992) found that lead uptake from garden plots in England was best predicted by total soil lead. In contrast, Zurera-Cosano et al. (1989) found no statistically significant correlation between the concentration of lead in soils and the amounts taken up by vegetables.

In contrast to uncontaminated soils, lead content in contaminated soils is usually higher in surface horizons than deeper in the soil profile. This trend is due in part to aerial deposition of lead as well as its tendency to form strong complexes with organic matter (Adriano 1986). In the presence of soluble organic acids, such as fulvic acid, lead can be mobilized and leach downward in a soil profile. Although the addition of organic matter to soil has been shown to decrease lead availability to plants, the eventual decomposition of the organic complexes may release the lead to the soil solution (Kabata-Pendias and Pendias 1992).

A number of techniques are available to decrease lead bioavailability. In addition to the effect of organic matter, manipulation of soil pH by liming or adding phosphate fertilizers have been shown to decrease the amount of uptake (Adriano 1986; Kabata-Pendias and Pendias 1992). In general, modifications of the soil that decrease the solubility of lead will decrease its availability to plants. Therefore, the liming of acidic soils to near-neutral pH as well as increasing the amount of  $\text{PO}_4$  in the soil system would cause a decrease in the bioavailability of lead. Unfortunately, very little research has been done on the speciation of lead in the soil solution and its effect on bioavailability.

### 3.2 Water

In aqueous systems the solubility of lead is affected by pH and by the concentrations of ligands that can form soluble complexes with the metal (Lindsay 1979). The amount of  $\text{Pb}^{2+}$  in solution increases by about two orders of magnitude with each unit of pH decrease if it stays in equilibrium with the common solid phases. Total lead in solution will be higher because of the formation of lead complexes with inorganic ligands such as  $\text{Cl}$ ,  $\text{OH}$ ,  $\text{F}$ , and  $\text{NO}_3$ , or with organic anions. Lead in surface waters may be undersaturated with respect to lead-containing minerals because of their scarcity in sediments and the kinetic limitations on dissolution reactions (Lindsay 1979).

In freshwater ecosystems, the toxicity of lead is reduced in hard water compared to soft water (Mance 1987). For example, in a 570-d experiment, the trout *Salmo gairdneri* developed blacktail in soft water containing a lead solution of  $12 \mu\text{g}\cdot\text{L}^{-1}$ , while there was no effect in hard water with  $200 \mu\text{g}\cdot\text{L}^{-1}$ . Long-term exposure of freshwater invertebrates indicate negative impacts with lead concentrations in the range of  $19\text{--}54 \mu\text{g}\cdot\text{L}^{-1}$ . Mance (1987) proposed a critical concentration of  $20 \mu\text{g}\cdot\text{L}^{-1}$  as the annual average lead concentration that would not cause negative effects on freshwater aquatic systems.

A review of the inputs of trace metals into the marine environment indicates that the majority of lead enters the oceans via the atmosphere. Chester and Murphy (1990) studied the chemistry of aerosols over the North Atlantic and indicated that the westerly winds contain lead from anthropogenic sources, while the northeast trade winds contain lead dust of crustal origin. Although some of the

atmospherically derived lead remains as insoluble particles that will sediment to the ocean floor, the proportion of atmospheric lead dissolving still exceeds the fluvial flux of lead to the marine environment.

Marine life seems to have a greater sensitivity to dissolved lead than freshwater species (Mance 1987). The most sensitive marine species were the larvae of molluscs and the zoeae of *Cancer magister*, which showed abnormal development after 2 d exposure to  $450 \mu\text{g}\cdot\text{L}^{-1}$ .

### 3.3 Air

Lead enters the atmosphere via several natural and anthropogenic processes, including wind erosion of soils and industrial materials, the burning of coal, wood, gasoline or sewage sludge, and the refining of lead and other metals (e.g., Cu, Zn, Ni). Even if lead is initially volatilized, it rapidly condenses or adheres to aerosols and other airborne particulates.

Dust containing particles of lead or lead compounds may travel great distances before being deposited back to earth. Distance travelled depends on the source of emission, the size of the particles, and atmospheric conditions. Lead from automobile exhaust tends to be deposited within several hundred metres of roadways, while lead from industrial refineries travels farther (several kilometres) because of its higher release into the atmosphere through smokestacks (Adriano 1986). Lead tends to form relatively small particles ( $0\text{--}2.5 \mu\text{m}$  in diameter) that may also be carried into the upper atmosphere and transported for hundreds or thousands of kilometres (Haygarth and Jones 1992).

## 4. BEHAVIOUR AND EFFECTS IN BIOTA

One way to assess the potential hazards of lead-contaminated soils to terrestrial organisms is to examine effects-based toxicity studies. Unfortunately, studies on the toxicity of lead to soil biota are based on the solid forms of the metal in soil; in many cases only total lead is measured, whereas soluble lead should also have been measured since there is considerable evidence suggesting that lead dissolved in the soil solution is directly linked to toxicity. Furthermore, lead in solution will exist as  $\text{Pb}^{2+}$  and as complexes with various ligands (e.g., Cl, OH,  $\text{SO}_4$ ,  $\text{CO}_3$ ,  $\text{HCO}_3$ , and organic ligands) that can affect the bioavailability of the metal to organisms (Morrison 1989). Therefore, the wide range of toxicities measured for organisms in contact with lead-contaminated soils may be due to metal variations in soil solution speciation. Until toxicity experiments are linked to detailed research on solution chemistry, it may be impossible to find the underlying mechanistic relationship between lead in soils and toxicity.

In general, the  $\text{Pb}^{2+}$  free ion can react directly with biological membranes and have a direct toxic effect. On the other hand, the bioavailability of heavy metals bound to organic or inorganic anions will depend on the strength of the bond. Strongly bound metals will tend to be nonbioavailable, while weakly bound metals may be readily taken up by biota. Lead molecules to lipid soluble complexes may be more bioavailable than those bound to other organic compounds (Morrison 1989).



The LOEC endpoints reported in the toxicity tables represent the lowest-observed-effects concentration at which there was a statistically and biologically significant difference from the controls, as reported by the author(s). If no such statistical tests were reported by the author(s), the percentage of adverse effect, as compared to the controls, resulting from lead concentrations within the soil will be calculated by the CCME from data presented by the author(s). This percentage of adverse effect is represented by an EC (effects concentration) endpoint within the toxicity tables. Actual EC<sub>xx</sub> endpoints reported by the author(s), such as EC<sub>25</sub> or EC<sub>50</sub>, will be presented as such without any calculation of a percentage of adverse effect. Measured concentrations and metal extraction methods are reported in the toxicity tables only if they involve digestion of soil with a strong acid, such as HCl or HNO<sub>3</sub>. Otherwise, the nominal concentrations are reported.

#### 4.1 Soil Microbial Processes

Toxicity studies on soil microbial processes that were selected for use in soil quality guidelines derivation are listed in Table 4. Soil enzyme activities were not included in this table since they may not represent measured effects of chemicals on soil microbial populations. This is because many enzymes produced by plants and microbes can exist and function extracellularly in soil for varying periods of time, depending on soil microenvironmental factors (Tabatabai 1982).

Bhuiya and Cornfield (1974) looked at the effects of PbO on nitrification and nitrogen mineralization in a 42-d incubation study of a sandy soil. They report that single doses of 1000 mg Pb·kg<sup>-1</sup> had no effect on nitrification at pH 6.0, but did inhibit nitrification by 11% and 9% at pH 7.0 and 7.7, respectively. The same study showed that nitrogen mineralization activity was increased slightly by the addition of 1000 mg Pb·kg<sup>-1</sup> at pH 6.0 and 7.0. However, at pH 7.7, nitrogen mineralization was reduced by 27%.

Liang and Tabatabai (1978) examined nitrification inhibition by lead in a 10-d incubation experiment for three soils with different composition and pH levels: soil #1 had a clay-silt-sand composition of 23%-39%-38% and a pH of 5.8; soil #2 had a composition of 30%-44%-26% and a pH of 7.8; and soil #3 had a composition of 34%-50%-16% with a pH of 7.4. Single-dose applications of lead acetate at 5000 µmole Pb·kg<sup>-1</sup>, or 1036 mg Pb·kg<sup>-1</sup>, reduced nitrification by 26%, 10%, and 7%, respectively, for soil #1, #2, and #3.

Bollag and Barabasz (1979) studied the nitrogen cycle in an autoclaved silt loam. After an incubation period of 21 d, the addition of Pb(NO<sub>3</sub>)<sub>2</sub> at a concentration of 1000 mg Pb·kg<sup>-1</sup> reduced denitrification by approximately 15% from controls (estimated from graphs presented by the authors) while no reduction was observed at 500 mg Pb·kg<sup>-1</sup>.

Wilke (1989) studied the effects of PbCl<sub>2</sub> on nitrification and nitrogen mineralization in a 28-d experiment. Nitrification was not inhibited at levels of 1000 and 4000 mg Pb·kg<sup>-1</sup>, but was actually increased by 12% and 16%, respectively. Nitrogen mineralization was reduced by 32% and 44% at concentrations of 1000 and 4000 mg Pb·kg<sup>-1</sup>, respectively.

The effect of  $\text{PbCl}_2$  on the respiration of two sandy soils and a clay soil was examined in a 42-h experiment by Doelman and Haanstra (1979). Respiration activity was measured in terms of  $\text{O}_2$  consumption. Decreases in  $\text{O}_2$  consumption occurred at concentrations of  $375 \text{ mg Pb}\cdot\text{kg}^{-1}$  and higher for both sandy soils: the first sandy soil showed decreases of 17%, 35%, 45%, and 67% at 375, 750, 1500, and  $3000 \text{ mg Pb}\cdot\text{kg}^{-1}$ , respectively, while decreases in the second sandy soil were 16%, 11%, 15%, and 84% at the same respective concentrations. Respiration was not affected in the clay soil at a concentration of  $375 \text{ mg Pb}\cdot\text{kg}^{-1}$ , but was reduced by 7%, 13%, and 35% at concentrations of 750, 1500, and  $3000 \text{ mg Pb}\cdot\text{kg}^{-1}$ , respectively.

In a later study, Doelman and Haanstra (1984) examined the effects of  $\text{PbCl}_2$  on soil respiration activity measured in terms of  $\text{CO}_2$  release. Four different soils were studied for various periods: a sandy soil (70 weeks), a sandy loam (43 weeks), a silty loam (90 weeks), and a clay soil (80 weeks). Decreases of 12%–59% were observed over a concentration range of 400–8000  $\text{mg Pb}\cdot\text{kg}^{-1}$  for the sandy soil. For the sandy loam, decreases of 6%–45% were reported over a range of 150–1000  $\text{mg Pb}\cdot\text{kg}^{-1}$ . No consistent pattern of respiration reduction in the silty loam was observed over the entire concentration range of 0–8000  $\text{mg Pb}\cdot\text{kg}^{-1}$  for the silty loam. Only small decreases (7%–8%) were noted over the range of 400–8000  $\text{mg Pb}\cdot\text{kg}^{-1}$ , and stimulation of respiration actually occurred at concentrations of 150 and 3000  $\text{mg Pb}\cdot\text{kg}^{-1}$ .

## 4.2 Terrestrial Invertebrates

### *Metabolic Fate and Behaviour*

Earthworms are important organisms in the soil macrofauna since they maintain aeration, water permeability, and mineral turnover in soil (Honda et al. 1984). Earthworms accumulate lead and are thus useful bioindicators of soil lead pollution. Total lead concentrations in soils almost always exceed the total lead concentrations in earthworms except where unique conditions, such as high levels of lead in soils combined with low pH and low calcium, cause earthworms to accumulate greater amounts of lead from the soil (Ireland 1979). Thus, bioconcentration factors expressing the ratio of lead in worms to lead in the soil range from 0.01 to 2.73, but are usually well below 1.0, indicating that there is no constant relationship between the concentration of lead in soil and that found in earthworms (Kabata-Pendias and Pendias 1992). Therefore, soil characteristics play a significant role in the uptake of lead by worms.

Ma (1982) found that the amount of lead in the earthworm *Aporrectodea caliginosa* was related to lead concentrations in soil since worms on contaminated soils had higher lead concentrations. However, physical and chemical factors such as pH, calcium content, and organic matter content in soil also affect the amount of lead accumulated by earthworms (Peramaki et al. 1992). At lower pH, soil adsorbs less lead, thus lead concentrations in the soil solution increase, rendering lead more bioavailable to earthworms (Kiewiet and Ma 1991). Lead accumulation in earthworms is usually higher in soils with low pH and low calcium concentrations and increases with decreasing organic matter content (Ma et al. 1983; Morgan and Morgan 1988). Since edaphic factors control the bioavailability of lead to earthworms, their lead body burdens provide a realistic indicator of the

environmental toxicity of this metal (Morgan and Morgan 1988).

Carnivorous soil invertebrates such as harvestmen and carabid beetles are generally more susceptible to lead poisoning than herbivores such as weevils and ants (Bengtsson and Rundgren 1984). Bioconcentration factors for lead in arthropods (i.e., ratio of lead concentration in animal to lead concentration in the litter layer) range from 0.01 to 0.43 (Martien and Hogervorst 1993). Like earthworms, there is no constant relationship between the amount of lead in the litter layer and the amount of lead accumulated by arthropods. Every species at a given site accumulates a different amount of lead, and the bioconcentration ratios of individual species vary between sites.

Stationary groups (ants and slugs) are more useful bioindicators than mobile ones (spiders and beetles). Lead levels in the body tissue of woodlice, *Porcellio scaber*, are positively correlated with lead levels in the leaf litter and soil (Hopkin et al. 1986). At contaminated sites, the lead concentrations of soft-bodied invertebrates may be elevated by lead dust attached to the body surface.

Invertebrates assimilate low net amounts of lead compared with other trace metals such as cadmium. Two processes limiting lead uptake in invertebrates are the rapid excretion of lead (van Straalen and van Meerendonk 1987) and its restricted absorption through the gut wall (Hopkin and Martin 1984).

All invertebrates cycle lead from the soil through their bodies. The chemical form of lead may be altered when it is excreted. Half of the body burden of lead in the collembola *Orchesella cincta* is in the gut, where it has a short half-life of less than a day. The flux of lead through consumption and defecation in *O. cincta* is about three orders of magnitude greater than the transfer of lead through the food chain (van Straalen et al. 1985). The collembola may therefore have a long-term solubilizing effect on lead in soils (van Straalen and van Meerendonk 1987).

### Toxicity

Invertebrate toxicity studies selected for use in soil quality guidelines derivation are presented in Table 5, while additional invertebrate studies that were consulted but not used in guidelines derivation are presented in Table 6.

Roberts and Dorough (1984) examined the effect of various chemicals on the earthworm *Eisenia fetida* for the purpose of using this organism as a possible indicator species for the relative toxicities of chemicals to soil invertebrates. The worms were exposed to deposits of various compounds on filter paper with mortality being recorded after 48 h. Lead nitrate [ $\text{Pb}(\text{NO}_3)_2$ ] was classified as "very toxic" with an  $\text{LC}_{50}$  range of 10–100  $\mu\text{g Pb}\cdot\text{cm}^{-1}$ , which is below the "super toxic" ( $<1 \mu\text{g}\cdot\text{cm}^{-1}$ ) and "extremely toxic" (1–10  $\mu\text{g}\cdot\text{cm}^{-1}$ ) classes, and above the "moderately toxic" (100–1000  $\mu\text{g}\cdot\text{cm}^{-1}$ ) and "relatively toxic" ( $>1000 \mu\text{g}\cdot\text{cm}^{-1}$ ) classes. Cadmium chloride and copper sulfate, which were the only other heavy metals tested, were ranked in the same class as lead.

Environment Canada (1995) reported a 14-d  $\text{LC}_{25}$ ,  $\text{LC}_{50}$ , and  $\text{LC}_{70}$  of 2067, 2500, and 3070  $\text{mg}\cdot\text{kg}^{-1}$ , respectively, for the earthworm *E. fetida* in artificial soil (7.1% organic matter, pH 4.3). A 14-d NOEC (no-observed-effect concentration) was reported at 1480  $\text{mg Pb}\cdot\text{kg}^{-1}$ .

Malecki et al. (1982) looked at the toxic effects of different lead compounds (lead acetate, lead carbonate, lead chloride, lead nitrate, and lead sulfate) on the growth and reproduction of young earthworms (*E. fetida*) over 8 weeks. The metals were mixed with 20 g of horse manure and placed on top of 30 g of soil. For all the lead compounds tested, significant adverse effects on reproduction occurred at lower concentrations (4000–14 000 mg Pb·kg<sup>-1</sup>) than those for growth (12 000–40 000 mg Pb·kg<sup>-1</sup>). The range of concentrations at which significant effects occurred indicate that the toxic effects of lead depend on the chemical form used and that high concentrations of lead are required to impair growth and reproduction of *E. fetida*. Ma (1982) found, however, that while 1000 mg Pb·kg<sup>-1</sup> did not affect the growth of adult *Lumbricus rubellus*, 3000 mg Pb·kg<sup>-1</sup> reduced their body weight by 41% after 12 weeks.

Several invertebrate studies compare the varying toxicities of different metals, such as cadmium, copper, nickel, zinc, and lead. Of the various heavy metals evaluated, lead is often reported as the least toxic. In a 6-week LC<sub>50</sub> study, lead was reported as the least toxic metal tested and ranked last in order of decreasing toxicity as follows: cadmium > copper > nickel > lead (Ma 1982). Spurgeon et al. (1994) examined the effects of lead, copper, cadmium, and zinc on the reproduction and survival of *E. fetida*. The mortality results for lead were a 14-d LC<sub>50</sub> of 4480 mg Pb·kg<sup>-1</sup>, a 56-d LC<sub>50</sub> of 3760 mg Pb·kg<sup>-1</sup>, and a 56-d NOEC of 2190 mg Pb·kg<sup>-1</sup>. For cocoon production, an EC<sub>50</sub> of 1940 mg Pb·kg<sup>-1</sup> and a NOEC of 1810 mg Pb·kg<sup>-1</sup> were reported. Cocoon production was more sensitive to lead toxicity than overall earthworm survival, but there was no significant effect on the viability of cocoons. Earthworms were found to be more tolerant of lead than either copper or cadmium for both survival and cocoon production, but they were more tolerant of zinc than lead.

Bengtsson et al. (1986) looked at lead toxicity to the earthworm *Dendrobaena rubida* at different pH (4.5, 5.5, and 6.5). Solutions of Pb(NO<sub>3</sub>)<sub>2</sub> were added to a soil composed of sand-sized particles from the C horizon of a coniferous forest soil and well-decomposed cattle dung. Soil pH was controlled by adding either H<sub>2</sub>SO<sub>4</sub> or CaCO<sub>3</sub>. No worms from the parent generation survived 4 months in the lead-contaminated soils of pH 4.5. In the reproduction portion of the study, the total number of earthworm hatchlings in pH 4.5 and 5.5 decreased by 11% and 22%, respectively, at 100 mg Pb·kg<sup>-1</sup>. In pH 6.5, however, cocoon production was higher in soils with 100 mg Pb·kg<sup>-1</sup> than in the uncontaminated soils.

### 4.3 Terrestrial Plants

#### *Metabolic Fate and Behaviour*

The behaviour of lead in terrestrial plants is of concern due to lead's potential phytotoxicity and its potential to contaminate the food chain. Plants are exposed to lead from soil and aerosol sources. The burden that lead puts on a plant may be a combination of uptake from roots and foliar uptake from airborne lead (Haygarth and Jones 1992). The availability of lead from the soil to terrestrial plants is a function of soil physical and chemical characteristics and plant biological characteristics (Bahlsberg 1989; Adriano 1986; Jones 1991; Tyler et al. 1989; Koeppe 1981; OMOE 1993). Uptake and distribution of lead in higher plants is influenced by the chemical form of lead, the presence of

other metal ions, the type of soil, and soil properties such as pH, cation exchange capacity (CEC), specific surface area, and organic matter content. Plant species, intraspecific variations, growth rate, developmental stage, and growth conditions such as temperature, light, and nutrient availability are all contributing factors to the interaction between lead and plants.

Lead is considered a nonessential element to plants, although certain studies have reported a stimulation effect on growth at low concentrations (Nakos 1979; Muramoto et al. 1990; Balba et al. 1991). In general, for most studies, significant adverse effects on plants were seen only at relatively high lead concentrations (Pahlsson 1989). Visible symptoms of lead toxicity include smaller leaves, chlorotic and reddish leaves with necrosis, short black roots, and stunted growth (Pahlsson 1989). In addition, exposed plants generally exhibit decreasing photosynthetic and transpiration rates with increasing lead concentrations. The responses are suggested to be related to changes in resistance of the stomata to CO<sub>2</sub> and diffusion of water (Bazzaz et al. 1974). Lead ions are also shown to inhibit chlorophyll biosynthesis leading to lowered chlorophyll contents. Thus, decreased photosynthesis could be partly related to reduced chlorophyll contents of leaves (Pahlsson 1989).

Phytotoxicity of lead has been manifested as reduced photosynthesis in loblolly pine (*Pinus taeda*) and red spruce (*Picea rubens*) (Seiler and Paganelli 1987). Reduced dry weight of roots and shoots also occurred due to excess lead in the two previous species (Seiler and Paganelli 1987), in radish (*Raphanus sativa*) (Khan and Frankland 1983), in tomato (*Lycopersicum esculentum*) and eggplant (*Solanum melongena*) (Khan and Khan 1983), and in corn (Miller et al. 1977). Balba et al. (1991) showed that reduced dry weight of tomato fruit was a result of lead toxicity. Ectomycorrhizal colonization may also be affected by lead (Chappelka et al. 1991; Bahlsberg 1989). A significant decrease in plant nutrient (i.e., Mn, Zn, Fe, Cu, and Na) concentrations was reported by Khan and Khan (1983) in tomato and eggplant studies.

Some vascular plants have developed mechanisms that enable them to tolerate high levels of metals, including lead. Some of the mechanisms discussed by Tyler et al. (1989) are avoidance (i.e., root redistribution), exclusion (i.e., selective uptake and selective translocation to protect sensitive organs), immobilization at the cell wall, and excretion (i.e., guttation, foliar leakage). Sieghardt (1990) found *Silene vulgaris* and *Minuartia verna* growing well on mining dump material with lead concentrations of 24 700 mg·kg<sup>-1</sup>.

Uptake and accumulation rates of lead vary among and within species and appears to be influenced to a greater extent by pH than by any other soil properties. Seiler and Paganelli (1987) reported markedly elevated lead toxicity for red spruce (*P. rubens*) due to the increased bioavailability of lead created from low pH conditions. Allinson and Dzialo (1981) found that ryegrass (*Lolium hybridum*) and oats (*Avena sativa*) contained significantly higher lead concentrations after 3 months of growth in a soil with pH 4.5 than in a second soil with pH 6.4. It is difficult to compare the bioconcentration factors (BCF) from different studies since each BCF is influenced by the choice of plant species, growth stage, growing conditions, soil, and form of lead used (Sheppard and Evenden 1988 and 1990; Sheppard and Sheppard 1991). Tobacco (*Nicotiana tabacum*) leaves accumulate between 24% to 49% of total soil lead concentrations (Adamu et al. 1989). Leita et al. (1989) measured lead content of plants growing on lead-contaminated sites and found fodder plants (Graminae) with

concentrations of 5.5%–174% of the total soil concentration and herbaceous species *Helichrysum italica* and *Scolymus hispanicus* with concentrations of 1.3%–51% of the total soil concentration. In a study of vegetables grown on soil with a history of sludge application, Chumbley and Unwin (1982) found that spinach (*Spinacia oleracea*) was the greatest accumulator of lead, with a bioconcentration factor of 0.03. Lead bioconcentration factors for most plants typically range from 0.001 to 0.03 (Jones and Johnston 1991). The OMOE (1992) adopted a general soil-to-plant bioconcentration factor of 0.039 for common backyard fruits and vegetables.

Plant roots are generally able to take up and accumulate large quantities of lead in soil and culture solutions, but translocation to aerial shoots and leaves is limited due to binding of lead ions (e.g., mainly  $Pb^{2+}$ ) at root surfaces and cell walls (Pahlsson 1989). Deposition of airborne lead onto plants is also an important contributor to total lead concentration in plants. Dalenberg and van Driel (1990) conducted an experiment designed to distinguish airborne lead from soil lead as a source for plants. The study indicated that a significant portion (75%–95%) of the lead found in the test plants was from atmospheric deposition. From a similar study, Tjell and Mosbæk (1979) concluded that aerial deposition of lead onto grass (*Lolium multiflorum*) was responsible for between 90% to 99% of whole plant lead concentrations. Jones and Johnston (1991) examined data from experimental grassland plots in southeast England since the mid-1800s and found evidence that atmospheric lead predominates over soil-derived lead in crop plants and grasses. There was also evidence to show that reduction in the amount of lead added to gasoline in England at the end of 1985 resulted in an overall decline in plant lead concentrations at rural locations.

The interaction of lead with other ions is important when considering trace metal phytotoxicity. Lead is rarely the only threat in a contaminated soil system. Kabata-Pendias and Pendias (1992) described the antagonistic effects of lead on phosphorus, calcium, and sulfur uptake and the possible antagonistic effects of zinc on lead uptake. They also discuss the synergistic effect of lead on cadmium uptake. Hassett et al. (1976) reported that reduced root elongation of corn (*Zea mays*) was greater when Cd and Pb were added together. Miller et al. (1977) reported increased cadmium concentrations in corn plants when lead was present in the system. In the same study, the effect of cadmium on lead uptake was quite different, as cadmium reduced the amount of lead taken up by corn plants. Allinson and Dzialo (1981) observed enhanced cadmium concentrations in ryegrass (*L. hybridum*) when lead was present in the system. They also observed depressed lead concentrations in ryegrass and oats (*A. sativa*) in the presence of cadmium.

### Toxicity

Plant toxicity studies selected for use in soil quality guidelines derivation are presented in Table 5, while additional plant studies that were consulted but not used in guidelines derivation are presented in Table 6.

In a 5-d experiment, Hassett et al. (1976) grew germinated corn seeds (*Z. mays*) in a loamy sand (pH 6.5) spiked with  $PbCl_2$  at concentrations from 0 to 2000 mg  $Pb \cdot kg^{-1}$ . No effect was observed at 100 mg  $Pb \cdot kg^{-1}$ . A significant reduction of 19% in root elongation occurred at 250 mg  $Pb \cdot kg^{-1}$ . Root elongation ceased completely at 2000 mg  $Pb \cdot kg^{-1}$ . Miller et al. (1977) looked at dry shoot

weight reduction in corn plants (*Z. mays*) exposed for different periods to a loamy sand with pH 6.0 spiked with  $\text{PbCl}_2$  at concentrations of 0, 125, and 250  $\text{mg Pb}\cdot\text{kg}^{-1}$ . After 17, 24, and 31 d of growth, significant reductions of 18%, 29%, and 13%, respectively, occurred at 125  $\text{mg Pb}\cdot\text{kg}^{-1}$ , but no significant reductions occurred during the 10-d test period.

Environment Canada (1995) reported 3-d results for NOEC, LOEC,  $\text{EC}_{25}$ , and  $\text{EC}_{50}$  toxicity endpoints as 421, 974, 833, and 1236  $\text{mg Pb}\cdot\text{kg}^{-1}$ , respectively, for radish (*R. sativa*) seedling emergence. In addition, a 5-d study gave NOEC, LOEC,  $\text{EC}_{25}$ , and  $\text{EC}_{50}$  values of 416, 740, 667, and 876  $\text{mg Pb}\cdot\text{kg}^{-1}$ , respectively, for seedling emergence of lettuce (*Lactuca sativa*). Both species were grown on artificial soil (OM 6.3%, pH 4.1).

Khan and Frankland (1984) exposed oats (*A. sativa*) for 42 d to a loamy sand (pH 5.4) spiked with  $\text{PbCl}_2$  at concentrations of 100, 500, and 1000  $\text{mg Pb}\cdot\text{kg}^{-1}$  soil. No significant effects were observed on root biomass at 100  $\text{mg Pb}\cdot\text{kg}^{-1}$ , but a significant reduction of 37% was reported at 500  $\text{mg Pb}\cdot\text{kg}^{-1}$ . Root biomass of wheat (*Triticum aestiva*) was also not affected in any significant manner at 100  $\text{mg Pb}\cdot\text{kg}^{-1}$ , but was significantly reduced by 15% at 500  $\text{mg Pb}\cdot\text{kg}^{-1}$ .

Allinson and Dzialo (1981) found that dry matter and seed yield of ryegrass (*L. hybridum*) were not only unaffected by a lead concentration of 250  $\text{mg Pb}\cdot\text{kg}^{-1}$ , but exhibited a significant increase. For oats (*A. sativa*), lead had no effect on dry matter yield, but significantly increased seed yield at 250  $\text{mg Pb}\cdot\text{kg}^{-1}$ . This study was performed in a fine sandy loam at two test pH levels (4.5 and 6.4) with  $\text{Pb}(\text{NO}_3)_2$  applied at 0 and 250  $\text{mg Pb}\cdot\text{kg}^{-1}$ . The same results for both plants were reported at both pH levels.

Dang et al. (1990) measured the growth of onion (*Allium cepa*) and fenugreek (*Trigonella poenumgraceum*) exposed to  $\text{Pb}(\text{NO}_3)_2$  concentrations of 0–400  $\text{mg Pb}\cdot\text{kg}^{-1}$  in soil. Germinated onion and fenugreek seedlings were planted in a clay loam (pH 8.3) until maturity. A significant reduction of 11% in dry weight yield of onions was observed at 50  $\text{mg Pb}\cdot\text{kg}^{-1}$ , while fenugreek required a concentration of 400  $\text{mg Pb}\cdot\text{kg}^{-1}$  to show a significant dry weight yield reduction of 20%. No significant effects were observed for fenugreek at the 200  $\text{mg Pb}\cdot\text{kg}^{-1}$  dose.

Khan and Khan (1983) observed a significant reduction of 25% in dry shoot weight for tomato (*L. esculentum*) and 42% for eggplant (*S. melongena*) at 600  $\text{mg Pb}\cdot\text{kg}^{-1}$  in a fine sandy loam (pH 8.8, OM 0.4%), while no significant effect was observed at 300  $\text{mg Pb}\cdot\text{kg}^{-1}$ . Low concentrations of  $\text{Pb}(\text{NO}_3)_2$  were actually beneficial to both tomato and eggplant as dry shoot weight was increased by 18% and 10% for tomato and eggplant, respectively, at 75  $\text{mg Pb}\cdot\text{kg}^{-1}$ . In a similar study, Balba et al. (1991) observed reduced tomato fruit dry weight over a range of  $\text{Pb}(\text{NO}_3)_2$  concentrations in a sandy soil (pH 8.2, OM 0.9%) and a clay soil (pH 7.5, OM 2.1%). In the sandy soil, reductions of 2%–39% in fruit dry weight yield were reported over a concentration range of 25–200  $\text{mg Pb}\cdot\text{kg}^{-1}$ . In the clay soil, reductions were from 21% to 40% in dry fruit weight over a range of 50–200  $\text{mg Pb}\cdot\text{kg}^{-1}$ , while a slight increase occurred at 25  $\text{mg Pb}\cdot\text{kg}^{-1}$ .

Seiler and Paganelli (1987) grew loblolly pine (*P. taeda*) and red spruce (*P. rubens*) seedlings in their native soils spiked with  $\text{PbCl}_2$  at concentrations of 0–1200  $\text{mg Pb}\cdot\text{kg}^{-1}$  for 19 weeks. Red spruce

appeared to be the more sensitive of the two species, although the lower pH conditions under which the red spruce was grown may have contributed to its greater sensitivity to lead. A significant reduction of 38%–45% in red spruce root and shoot dry weight and in plant height, respectively, occurred at  $150.1 \text{ mg Pb}\cdot\text{kg}^{-1}$ , while a significant reduction of 30% in photosynthesis was observed at  $271.1 \text{ mg Pb}\cdot\text{kg}^{-1}$ . Loblolly pine, on the other hand, displayed significantly reduced height (30% reduction) and dry weight of roots (61% reduction) and shoots (38% reduction) only at the highest concentration of  $1179 \text{ mg Pb}\cdot\text{kg}^{-1}$ . Photosynthesis was unaffected in loblolly pine even at the highest concentration.

Stunted growth was the effect that lead had on radish plants (*R. sativa*) (Khan and Frankland 1983). The effects of PbO and PbCl<sub>2</sub>, applied to a loamy sand (pH 5.4), were compared in a 42-d study. PbCl<sub>2</sub> was applied at rates of 0–5000 mg Pb·kg<sup>-1</sup> to the soil. Root growth was progressively inhibited once a soil concentration of 550 mg Pb·kg<sup>-1</sup> was reached, with an EC<sub>50</sub> of 2400 mg Pb·kg<sup>-1</sup> as calculated from a dose-response curve. Shoots were less sensitive and exhibited an EC<sub>50</sub> at 2800 mg Pb·kg<sup>-1</sup>, while plant growth ceased at 5000 mg Pb·kg<sup>-1</sup>. Stems were reported to be thin and short with small leaves at the highest concentration. The experiment was repeated with lead oxide (PbO) at concentrations of 0–10 000 mg Pb·kg<sup>-1</sup>. The EC<sub>50</sub> on shoot growth was extrapolated to 12 000 mg Pb·kg<sup>-1</sup>, while roots were, again, more sensitive with an EC<sub>50</sub> of 10 000 mg Pb·kg<sup>-1</sup>.

#### 4.4 Mammals and Birds

##### *Metabolic Fate and Behaviour*

Lead poses a threat to mammals and birds through a number of exposure routes. Mammals and birds inhale airborne lead directly or ingest particulate matter deposited on the ground and vegetation. Direct ingestion of contaminated soil and grit occurs when herbivores and birds feed and groom. Animals lick painted surfaces and drink lead-contaminated water. Waterfowl and raptors ingest substantial amounts of lead shot and fishing weights when they feed. Herbivores, insectivores, and carnivores at all trophic levels are exposed to lead by eating lead-contaminated vegetation or prey.

Lead toxicosis has been observed in many animals, but its effects are so diverse that it is difficult to identify any single organ failure as being responsible for death (Beyer et al. 1988; Humphreys 1991). Clinical signs of lead poisoning include behavioural aberrations such as vocalization, aggression, memory loss, muscle spasms, convulsions, imbalance, dehydration, emaciation, and impaction of the gastrointestinal tract (Morgan et al. 1975; MacDonald et al. 1983; O'Halloran and Myers 1988).

Lead ingested as lead acetate, lead phosphate, lead oxide, lead carbonate, or from paint, is absorbed more easily than lead ingested as the metal or lead sulfide (powdered galena ore) (Humphreys 1991). The bioavailability of lead ingested directly from soil is influenced by the source and speciation of the lead, the size of lead-bearing particles, and the concentration of other elements, such as copper and zinc ingested at the same time (Steele et al. 1990; Chaney et al. 1989). For example, the bioavailability of lead in mining waste soils is lower than that for lead ingested from paints or automotive exhausts because the solubility in mining soils is controlled by lead sulfide, which is



insoluble, and because concentrations of other metals (e.g., copper, zinc) are high, reducing lead absorption in the gut (Freeman et al. 1992).

Lead poisoning has been reported in all types of livestock and the common exposure routes are through leaded paints in old barns, machinery wastes such as oil and discarded batteries in farm yards and pastures, and direct ingestion of lead-contaminated soils and forage. Elevated lead in livestock is associated with aerially deposited lead from smelters and automotive sources (Koh and Judson 1986; Amodio-Cocchieri and Fiore 1987). A seasonal pattern of lead poisoning has been observed in cattle (Buck 1975). Increased photoperiod during summer months may increase vitamin-D, which increases the rate of lead absorption in the body. In addition, spring forage may induce hypophosphatemia, which increases pica-like behaviour. Ruminants are less sensitive to lead than monogastric animals. However, swine are generally more resistant to lead than cattle, goats, and sheep (Humphreys 1991).

In lead-contaminated areas, total lead concentration in surface soils is higher than in forage crops, thus direct ingestion of soil is more likely to contribute to elevated lead levels in livestock than the consumption of forage grown on that soil (Zach and Mayo 1984; Brebner et al. 1993). Soil ingestion rates decrease when vegetation is lush, and increase with the proportion of bare ground (McMurter 1993).

Neuman and Dollhopf (1992) measured concentrations of lead in whole blood of 222 cattle from nine herds. Lead levels in blood were significantly elevated for herds near a primary lead smelter when compared to other herds. Herd blood lead levels were significantly correlated with the distance from the smelter and with soil lead concentrations. Immature cattle (less than 1 year old) had significantly higher blood lead levels than mature cattle. Neuman and Dollhopf (1992) also suggested that soil may be more important than forage as a source of lead for livestock.

Small mammals exposed to lead accumulate the metal in the renal and liver tissues and have a decreased body weight-to-organ ratio (Goyer et al. 1970). Most studies show that the accumulation of lead in ground-living mammals is highest in areas of elevated lead contamination. Within an ecosystem, herbivorous and carnivorous small mammals are used as bioindicators of lead contamination because they show the extent to which lead in that environment enters the food chain either via plant root uptake or via soil fauna (Ma 1989). Lead accumulation in secondary insectivores such as shrews and moles is greater than for herbivorous rodents. For example, Trowbridge's shrews (*Sorex trowbridgii*) and shrew-moles (*Neurotrichus gibbsii*) collected in a sludge-treated forest in Washington state accumulated more lead than granivorous deer mice (*Peromyscus maniculatus*) (Hegstrom and West 1989). Kidney concentrations in all shrews (*Sorex araneus*) and 24% of bank voles (*Clethrionomys glareolus*) on a shooting range in the Netherlands exceeded  $25 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw}$ , while those of the wood mice (*Apodemus sylvaticus*) remained lower (Ma 1989). Even on highly contaminated soil, the amount of lead in house mice (*Mus musculus*) from disposal sites for dredged material (Maryland, USA) was low, indicating that the lead did not enter the food chain of the mouse (Beyer et al. 1990). In general, shrews and moles are more sensitive indicators of lead toxicity than other small mammals because a large proportion of their diet consists of earthworms, which are known to accumulate lead (Southern 1964; Ma 1987).

The bioavailability of lead from contaminated soil is only partly dependent on the total amount of lead in the soil. In the Netherlands, the amount of lead in moles (*Talpa europaea*) from heathland sites was 18 times higher in the kidney and 4 times higher in the liver, than moles from pasture sites with similar soil lead concentrations (Ma 1987). The accumulation of lead in moles may reflect the differing bioavailability of lead to the earthworms of the two sites. Earthworms from the acidic sandy soils of the heathlands accumulated up to 10 times more lead than those on the limed pasture soils.

### Toxicity

#### Mammals

Generally, effects studies using livestock and small mammals are lacking, and laboratory rodent studies concentrate on endpoints such as behavioural toxicity, which have a direct relevance to lead toxicosis observed in humans. Evidence suggests that high concentrations of lead will inhibit reproduction and increase mortality of offspring, but reliable toxicity studies that focus on the survival of mammalian species are currently lacking. Mammalian oral toxicity studies for lead that were selected for use in soil quality guidelines derivation are presented in Table 7.

Lassen and Buck (1979) investigated the toxicity of lead to swine by giving 6-week-old pigs lead acetate in water, either orally ( $0\text{--}35.2\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}$ ) or intraperitoneally ( $0\text{--}8.8\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}$ ) on a daily basis. The doses were administered 6 d each week for 13 weeks. None of the pigs given oral doses of lead died, but clinical signs of lead toxicosis (coughing, rough coats, gaunt appearance) were seen in two of the three pigs treated at  $35.2\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}$ . All the pigs given intraperitoneal doses of Pb acetate had clinical signs of lead toxicosis. In addition, two of the three pigs treated at  $4.4\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}$  died on days 25 and 35, and two of the three pigs on  $8.8\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}$  died on days 23 and 38. Fick et al. (1976) fed lead acetate ( $0\text{--}1000\text{ mg Pb}\cdot\text{kg}^{-1}$  of feed) to 20 wether lambs (36 kg) for 84 d and found that the highest dose, translating to  $44.4\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$ , did not affect feed consumption rates or weight gain.

Lynch et al. (1976) administered lead as  $\text{PbCO}_3$  in gelatin capsules to Holstein steer calves. The doses were 9 and  $18\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}$  3 times a week (the daily equivalent of  $3.9$  and  $7.7\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$ , respectively) for 12 weeks. A 13% reduction in weight was seen in calves receiving  $7.7\text{ mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$ , while only a 6% reduction was observed in calves receiving the lower dose. Both doses of lead significantly decreased erythrocyte hemoglobin concentration, reduced erythrocyte acid dehydrogenase (ALAD) activity, and increased the percentage of lymphocytes in blood. One of the known specific effects of lead has been the reduction of hemoglobin synthesis (Hemberg et al. 1970). Part of this effect is produced by inhibition of the erythrocyte ALAD enzyme.

The effects of lead on reproduction in livestock are not widely reported. However, a study on adult buffaloes and cows in an industrialized region of the Punjab state in India reported that lead concentrations in maternal blood, umbilical cord blood, amniotic fluid, and placenta in animals experiencing fetal death were significantly higher than those with normal parturition (Kaur 1989).

In an attempt to bridge the gap between clinical laboratory effects studies, which use rodents from genetically controlled stocks, and studies on species in the wild, Zakrzewska (1988) looked at the effects of lead on the postnatal development of bank voles (*Clethrionomys glareolus*) collected from a forest in Poland. After parturition, female bank voles were fed wheat containing  $\text{Pb}(\text{NO}_3)_2$  at 60, 380, and 730  $\text{mg Pb}\cdot\text{kg}^{-1}$ . Growth rates and mortality were monitored for 20 d in the offspring, which fed exclusively on mother's milk for the first 15 d and subsequently on milk and contaminated food for the next 5 d. The growth of young voles feeding in the 730  $\text{mg Pb}\cdot\text{kg}^{-1}$  group was inhibited. Mortality was also highest in this group, but factors other than lead, like cannibalism, may have contributed to death in young bank voles.

Laboratory rodent studies are not recommended for use in the derivation of soil quality guidelines when acceptable livestock and wildlife studies are available, but they give more insight into the toxic effects of lead on the survival of mammals. Barratt et al. (1989) gave male rats lead acetate in drinking water (0.3, 33, and 330  $\text{mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$ ) for 9 weeks. A dose of 0.3  $\text{mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$  significantly reduced the number of spermatozoa within the cauda, while higher doses did not. The authors suggest that lead probably affects the sperm after its release from the germinal epithelium and may increase the natural mechanism of sperm resorption. Rats treated at 330  $\text{mg}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$  had a significant reduction of 29% in body weight and prostate weight, and produced a higher percentage of abnormal spermatozoa than control animals. Murthy et al. (1991) gave male rats lead acetate in drinking water (4.4  $\text{mg Pb}\cdot\text{rat}^{-1}\cdot\text{d}^{-1}$  for 30 d followed by 5.2  $\text{mg Pb}\cdot\text{rat}^{-1}\cdot\text{d}^{-1}$  for 40 d) during testicular maturation before the prepubertal stage of development. This dose did not cause a change in the body weight of rats, but did result in a significant accumulation of lead in the testes.

Donald et al. (1986) administered a 0.13% lead acetate solution as drinking fluid to male and female mice for 20 d before breeding, during gestation and weaning, and to the pups after weaning. This dose of lead had no significant effect on the reproductive success of the mice or the birthweight of the offspring, but did delay the growth of fur and reduced the amount of weight gained in both male and female pups. Water intake at 28 weeks (adulthood) was not significantly affected by lead; males consumed 112  $\text{mL}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$  and females 93  $\text{mL}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$ . Offspring exposed to lead during gestation and weaning showed more behavioural changes, such as enhanced social and sexual investigation, than pups exposed to the same dose postnatally.

Freeman et al. (1992) fed Sprague-Dawley rats (7–8 weeks old) either a diet containing 0.2%–5% mining waste soil (3908  $\text{mg Pb}\cdot\text{kg}^{-1}$ , pH 3.7) or feed spiked with lead acetate at 1–250  $\text{mg Pb}\cdot\text{kg}^{-1}$  feed. The highest lead doses of both diets did not affect growth, food consumption, or liver weight in the rats. These doses corresponded to 12.1 and 23.2  $\text{mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$  for males and females, respectively, on the mining waste soil diet, and 16.3 and 25.7  $\text{mg Pb}\cdot\text{kg}^{-1}\text{ bw}\cdot\text{d}^{-1}$  for males and females on the lead acetate diet. Even though there were no differences in the toxic effects of the two diets, rats on the Pb acetate diet accumulated significantly higher levels of lead in the blood, bone, and liver than those on the soil diet, indicating that lead in the mining waste soil is less bioavailable than the salt form.

These studies indicate that exposure to lead can impair reproduction and growth in mammals. However, more toxicological effects-based studies must be carried out in order to establish which

exposure scenarios pose a real threat to the survival of different species.

## Birds

The endpoints investigated in various avian studies include growth rate, body weight, brain, liver and kidney weights, and effects on reproduction (e.g., egg production, eggshell thickness). A comparison of the studies in terms of daily lead intake per body weight of bird indicates that reproduction in precocial species (e.g., Japanese quail) is more sensitive to lead than altricial species (e.g., American kestrels). Avian oral toxicity studies for lead that were selected for use in soil quality guidelines derivation are presented in Table 7.

Metallic lead powder was mixed with corn oil and given orally to American kestrel (*Falco sparverius*) hatchlings for the first 10 d of life (Hoffman et al. 1985). After 5 d, the growth rate and weight gain of the hatchlings on  $125 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$  was significantly reduced by 16%. By day 6, 40% of the hatchlings on the  $625 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$  diet had died. The examination of 10-d-old hatchlings revealed that the hatchlings on  $125 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$  had lower brain weights and reduced crown rump and skeletal length, and those on  $625 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$  had lower liver and kidney weights than birds receiving no lead. In terms of lead concentrations in the diet, nestling American kestrels are more sensitive to lead than adult birds. This is because the diet-to-body-weight ratio is higher for nestlings than adults. On a  $25 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$  diet, nestlings will ingest more lead per body weight than adults since they consume 100% of their body weight in feed per day compared to about 21% for adults.

Custer et al. (1984) fed 1- to 2-year-old male and female American kestrels with a diet of biologically incorporated lead for 60 d. The diet consisted of homogenised 4-week-old cockerels raised on a poultry starter mash mixed with different concentrations of lead acetate. The highest dose of biologically incorporated lead,  $448 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ dw}$  of diet, translated to about  $28 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$  (assuming an average daily consumption of 25 g wet weight of chicken diet for a 120 g kestrel). At  $28 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$ , lead had no effect on the survival or body weight of kestrels and did not alter hematocrit, hemoglobin, or erythrocyte counts in blood, which are primary signs of lead toxicity. Biologically incorporated lead was probably less available than lead acetate, since most of the lead in the homogenized chicken feed was probably in bones and thus less available than lead in soft tissues and organs, such as the liver and kidneys. As the kestrels showed no signs of lead toxicity, poisoning in the wild is more likely to be caused by lead shot ingestion than biologically incorporated lead since the levels in the chicken diet ( $138 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ wet weight}$ ) were greater than those found in lead-poisoned birds. Furthermore, Stendell et al. (1989) reported that American kestrels fed pine voles containing  $38 \text{ mg Pb} \cdot \text{kg}^{-1}$  only accumulated sublethal amounts of lead ( $1 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ wet weight}$ ) in their livers.

Pattee (1984) added metallic lead (0, 10, and  $50 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ diet}$ ) to commercial bird of prey diet and fed it to paired American kestrels (1–6 years old) for 5–7 months until their clutch was completed. Food consumption was not reported, but assuming a kestrel weight of 130 g and an average feed intake of 5 g dry weight per day (Pattee 1984; P. Warrington 1995, British Columbia Ministry of the Environment, Lands, and Parks, pers. com.), the birds given the maximum concentration of

50 mg Pb·kg<sup>-1</sup> feed consumed lead at a rate of 1.9 mg Pb·kg<sup>-1</sup> bw·d<sup>-1</sup>. This dose had no effect on the initiation of egg laying, the size of the clutch, the percentage of fertile eggs laid, or the thickness of the eggshells, and therefore did not affect reproduction or survival of *F. sparverius*. Females accumulated more lead in bones and liver than males.

Edens and Garlich (1983) added lead acetate to the feed of both Hy-Line W-36 domestic leghorn chicken hens and to Japanese quail hens (*Coturnix coturnix japonica*). The quail hens, given the lead diets from the day of hatching, showed more susceptibility to the inhibitive property of lead on reproduction than the chicken hens who started the diets after 15 weeks of egg production. After 5 weeks, quail hens showed a significant decrease of 28% in egg production at 10 mg Pb·kg<sup>-1</sup> feed (or 1.8 mg Pb·kg<sup>-1</sup> bw·d<sup>-1</sup>), while a significant decrease of 7% was observed after 4 weeks at 50 mg Pb·kg<sup>-1</sup> feed (or 26.1 mg Pb·kg<sup>-1</sup> bw·d<sup>-1</sup>) for chickens. Lower calcium concentrations in the diets of quails (2.5%) compared to that for chickens (3.8%) may have enhanced the toxicity of lead to quails. The authors suggested that the decrease in egg production of Japanese quails is associated with reductions in plasma calcium.

The relationship between dietary lead, blood calcium, and egg production in birds is unclear. Birds are apparently different than some mammals in the manner in which they manage lead (Edens and Garlich 1983). In both birds and mammals, lead appears to be a bone-seeking element (Stone and Soares 1974). However, in rats, lead will cause the mobilization of bone calcium and, consequently, increased blood calcium. In birds, blood calcium declines after lead treatments (Yamamoto et al. 1974).

## 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 lead are based on the procedures described in CCME (1996).

All data selected for use in the following derivations have been screened for ecological relevance and are presented in Tables 4, 5, and 7. Invertebrate and plant toxicological studies that have been consulted but not used in guideline derivation are presented in Table 6. Studies were excluded from use because of one or more of the following reasons:

- soil pH was not recorded;
- soil pH was below 4 (as this is considered outside the normal pH range of most soils in Canada);
- 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 statistically significant according to the study from which the data were taken.

According to section 7.5.2.2 of the protocol (CCME 1996), the geometric mean should be used when multiple data are available for the same endpoint with the same species. For the lead data, the geometric mean has been applied to the LC<sub>50</sub> values for the earthworm *E. fetida* from Environment Canada (1995), Spurgeon et al. (1994) and Neuhauser et al. (1985), and the NOEC values for the earthworm *E. fetida* from Environment Canada (1995) and Spurgeon et al. (1994).

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

### 5.2.1 Soil Quality Guideline for Soil Contact (SQG<sub>SC</sub>)

The derivation of the soil quality guideline for soil contact (SQG<sub>SC</sub>) is based on toxicological data for vascular plants and soil invertebrates. The toxicological data for plants and invertebrates selected according to CCME (1996) are presented in Table 5. 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:

$$\text{TEC} = \text{NPER} / \text{UF}$$

where,

TEC = threshold effects concentration (mg·kg<sup>-1</sup> soil)

NPER = no potential effects range (25th percentile of effects and no effects data distribution) (mg·kg<sup>-1</sup> soil)

UF = uncertainty factor (if needed); no uncertainty factor was applied.

Out of a total of 47 data points, the 25th percentile corresponds to the 12th datum point of 250 mg Pb·kg<sup>-1</sup> soil from the Hassett et al. (1976) test on root elongation of corn (*Zea mays*).

Thus,

TEC = 250 mg·kg<sup>-1</sup> soil.

### *Nutrient and Energy Cycling Check*

The nutrient and energy cycling check was calculated using the selected microbial processes data presented in Table 4. Nitrification and nitrogen fixation data are considered to be primary data, whereas nitrogen mineralization, denitrification, and carbon cycling data are considered secondary data. LOEC data, as reported by the author, are used directly, while effective concentration (EC) data producing >15% and <40% effects in primary data (i.e., EC<sub>15</sub> to EC<sub>40</sub>) and >15% and <25% effects in secondary data (i.e., EC<sub>15</sub> to EC<sub>25</sub>) are interpreted as LOEC values. Insufficient primary data were available for the calculation, so the primary and secondary data were combined and the

check was carried out using a modified LOEC method whereby the geometric mean of available LOECs is calculated as the nutrient and energy cycling check.

The nutrient and energy cycling check (NECC) is calculated as follows:

$$\text{NECC} = (\text{LOEC}_1 \cdot \text{LOEC}_2 \cdot \text{LOEC}_3 \cdot \dots \cdot \text{LOEC}_n)^{1/n}$$

where,

NECC = nutrient and energy cycling check ( $\text{mg} \cdot \text{kg}^{-1}$  soil)

LOEC = lowest-observed-effects concentration ( $\text{mg} \cdot \text{kg}^{-1}$  soil)

n = number of available LOECs

Thus,

$$\begin{aligned} \text{NECC} &= (1036 \cdot 1036 \cdot 1036 \cdot 1000 \cdot 100 \cdot 3000 \cdot 400 \cdot 375 \cdot 375)^{1/9} \\ &= 643 \text{ mg} \cdot \text{kg}^{-1} \text{ soil} \end{aligned}$$

Since the TEC ( $250 \text{ mg} \cdot \text{kg}^{-1}$  soil) is lower than the NECC ( $643 \text{ mg} \cdot \text{kg}^{-1}$  soil), the TEC is considered to be protective of microbial nutrient and energy cycling processes and is adopted directly as the  $\text{SQG}_{\text{SC}}$  for agricultural and residential/parkland land uses.

### 5.2.2 Soil Quality Guidelines for Soil and Food Ingestion ( $\text{SQG}_I$ )

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

Calculation of the  $\text{SQG}_I$  is based on the lowest-observed-adverse-effects level (LOAEL) taken from the selected mammalian and avian toxicological data listed in Table 7. The LOAEL indicating the species most threatened was  $0.18 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$ , which resulted in a significant reduction in egg production for the Japanese quail (*Coturnix coturnix japonica*) (Edens et al. 1976).

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

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

where,

DTED = daily-threshold-effects dose ( $\text{mg} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$ )

LOAEL = lowest-observed-adverse-effects level ( $\text{mg} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$ )

UF = uncertainty factor; no uncertainty factor was applied

Thus,

$$\text{DTED} = 1.8 \text{ mg} \cdot \text{kg}^{-1} \text{ bw} \cdot \text{d}^{-1}$$

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:

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

where,

SIR = the soil ingestion rate ( $\text{kg dw soil} \cdot \text{d}^{-1}$ )

DMIR = geometric mean of available dry matter intake rates ( $\text{kg} \cdot \text{d}^{-1}$ ), which was determined to be  $0.027 \text{ kg} \cdot \text{d}^{-1}$  (P. Warrington 1995, British Columbia Ministry of the Environment, Lands, and Parks, pers. com.).

PSI = geometric mean of available soil ingestion proportions reported with the DMIR. As no information is available on the PSI for the species used, a default value of 0.077 (McMurter 1993) was used for the above equation.

Thus,

$$\text{SIR} = 0.027 \cdot 0.077 = 0.00208 \text{ kg dw soil} \cdot \text{d}^{-1}$$

The SIR can then be combined with the bioavailability factor (BF), body weight (BW), and a concentration of the contaminant in the soil ( $\text{SQG}_1$ ) to represent the exposure from soil ingestion. The soil concentration at this point is unknown, but it should not provide for more than 75% of the DTED when combined with the exposure calculated for food ingestion:

$$\text{exposure from soil ingestion} = \text{SIR} \cdot \text{BF} \cdot \text{SQG}_1 / \text{BW}$$

where,

SIR = soil ingestion rate ( $\text{kg dw soil} \cdot \text{d}^{-1}$ )

BF = bioavailability factor; because of lack of specific information on the bioavailability of lead from ingested soil for livestock and terrestrial wildlife, a BF of 1 is assumed (CCME 1996).

$\text{SQG}_1$  = concentration of the contaminant in soil that will not result in >75% DTED ( $\text{mg} \cdot \text{kg}^{-1}$  soil)

BW = mean body weight (kg); the mean body weight of Japanese quail was determined to be 0.152 kg (P. Warrington 1995, British Columbia Ministry of the Environment, Lands, and Parks, pers. com.).



### *Exposure from Food Ingestion*

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

$$\text{FIR} = \text{DMIR} - \text{SIR}$$

where,

FIR = food ingestion rate (kg dw food·d<sup>-1</sup>)

DMIR = geometric mean of dry matter intake rates (kg dw food·d<sup>-1</sup>)

SIR = soil ingestion rate (kg dw soil·d<sup>-1</sup>)

Thus,

$$\text{FIR} = 0.027 - 0.00208 = 0.0249 \text{ kg dw food} \cdot \text{d}^{-1}$$

The FIR can then be combined with the bioconcentration factor (BCF), the BW, and the SQG<sub>I</sub> to express the exposure from food ingestion:

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

where,

FIR = food ingestion rate (mg·kg<sup>-1</sup> dw food·d<sup>-1</sup>)

BCF = bioconcentration factor; estimated to be 0.035 from Jones and Johnston (1991) and OMOE (1992)

SQG<sub>I</sub> = concentration of the contaminant in soil that will not result in >75% DTED (mg Pb·kg<sup>-1</sup> soil)

BW = mean body weight (kg); the mean body weight of Japanese quail was determined to be 0.152 kg (P. Warrington 1995, British Columbia Ministry of the Environment, Lands, and Parks, pers. com.)

### *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<sub>I</sub>:

$$(\text{SIR} \cdot \text{BF} \cdot \text{SQG}_I / \text{BW}) + (\text{FIR} \cdot \text{BCF} \cdot \text{SQG}_I / \text{BW}) = 0.75 \text{ DTED}$$

$$\text{SQG}_I = (0.75 \text{ DTED} \cdot \text{BW}) / (\text{SIR} \cdot \text{BF} + \text{FIR} \cdot \text{BCF})$$

$$\text{SQG}_I = (0.75 \cdot 1.8 \cdot 0.152) / (0.00208 \cdot 1 + 0.0249 \cdot 0.035)$$

$$\text{SQG}_I = 70 \text{ mg Pb} \cdot \text{kg}^{-1} \text{ soil}$$

## 5.3 Soil Quality Guidelines for Commercial and Industrial Land Uses

### 5.3.1 Soil Quality Guidelines for Soil Contact (SQG<sub>sc</sub>)

The derivation of the SQG<sub>sc</sub> is also based on toxicological data for vascular plants and soil invertebrates presented in Table 5. However, for commercial and industrial land uses, only the effects data are used and uncertainty factors are not applied. There were 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·kg<sup>-1</sup> soil)

ERL = effects range low (25th percentile of effects data distribution) (mg·kg<sup>-1</sup> soil)

Out of a total of 24 data points, the 25th percentile corresponds to the 6th datum point of 400 mg·kg<sup>-1</sup> soil from the Dang et al. (1990) study on the reduction of dry weight of fenugreek (*T. poenumgraceum*).

Thus,

$$ECL = 400 \text{ mg} \cdot \text{kg}^{-1} \text{ soil.}$$

### *Nutrient and Energy Cycling Check*

The nutrient and energy cycling check was calculated using the selected microbial processes data presented in Table 4. Nitrification and nitrogen fixation data are considered to be primary data, whereas nitrogen mineralization, denitrification, and carbon cycling data are considered secondary data. LOEC data, as reported by the author, are used directly, while effective concentration (EC) data producing >15% and <50% effects in primary data (i.e., EC<sub>15</sub> to EC<sub>50</sub>) and >15% and <35% effects in secondary data (i.e., EC<sub>15</sub> to EC<sub>35</sub>) are interpreted as LOEC values. Insufficient primary data were available for the calculation, so the primary and secondary data were combined and the check was carried out using a modified LOEC method whereby the geometric mean of available LOECs is calculated as the nutrient and energy cycling check.

The nutrient and energy cycling check (NECC) is calculated as follows:

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

where,

NECC = effects concentration low (mg Pb·kg<sup>-1</sup> soil)

LOEC = lowest observed effects concentration (mg Pb·kg<sup>-1</sup> soil)

n = number of available LOECs

Thus,

$$\begin{aligned}\text{NECC} &= (1036 \cdot 1036 \cdot 1036 \cdot 1000 \cdot 100 \cdot 3000 \cdot 400 \cdot 375 \cdot 375 \cdot 1000 \cdot 1000 \cdot 3000)^{1/12} \\ &= 787 \text{ mg}\cdot\text{kg}^{-1} \text{ soil}\end{aligned}$$

Since the ECL (400 mg·kg<sup>-1</sup> soil) is lower than the NECC (787 mg·kg<sup>-1</sup> soil), the ECL is considered to be protective of microbial nutrient and energy cycling processes and is adopted directly as the SQG<sub>SC</sub> for commercial and industrial land uses.

#### 5.4 Derivation of the Final Environmental Soil Quality Guidelines (SQG<sub>E</sub>)

The following environmental soil quality guidelines are optimized for soils within the pH range of 4.0–8.8. The toxicological studies upon which these guidelines are based were conducted within this pH range. A summary of all the derived soil quality guidelines is presented in Table 8.

##### *Agricultural Land Use*

The lower value from the two procedures (SQG<sub>SC</sub> and SQG<sub>I</sub>) is selected as the final environmental soil quality guideline for agricultural land. The lower of the two procedures is the SQG<sub>I</sub>. Therefore, the final SQG<sub>E</sub> is 70 mg·kg<sup>-1</sup> soil.

##### *Residential/Parkland Land Use*

The SQG<sub>SC</sub> of 250 mg·kg<sup>-1</sup> soil is the final SQG<sub>E</sub> for residential/parkland land use.

##### *Commercial and Industrial Land Use*

The SQG<sub>SC</sub> of 400 mg·kg<sup>-1</sup> soil is the final SQG<sub>E</sub> for commercial and industrial land use.

## 6. DATA GAPS

Sufficient data exist on the toxicity of lead to soil ecosystem receptors to derive a soil quality criteria for the three major uses of land: agricultural, residential/parkland, and commercial/industrial land uses. Additional data are required, however, to refine our understanding of the fate and effects of lead in the environment.

A significant database exists on the fate of lead in soils and other environmental media. However, much of this information is inconclusive and these data are frequently contradictory. It is likely that a variety of factors, such as pH, moisture content, clay content, and organic matter content, act together to influence the fate of lead in soils. Additional information is required to understand fully the importance of each of these factors in determining the fate of anthropogenic lead in soils. These data will also contribute to our understanding of the bioavailability of lead in various soil types.

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## **TABLES**

**Table 1. Physical and Chemical Properties of Lead and Its Principle Compounds**

Form of Lead	Chemical formula	Properties	Molecular weight	Density (water = 1)	Melting point (°C)	Boiling point (°C)	Solubility (g·100 mL <sup>-1</sup> cold water)
Lead	Pb	Silvery-bluish grey, soft, cubic	207.19	11.34	327.5	1740	Insoluble
Lead acetate	Pb(C <sub>2</sub> H <sub>3</sub> O <sub>2</sub> ) <sub>2</sub>	White crystal	325.28	3.25	280	--	44.3
Lead carbonate	PbCO <sub>3</sub> , Natural cerussite	Colorless, rhombic	267.20	6.6	315	--	0.000 11
Lead chloride	PbCl <sub>2</sub>	White, rhombic	278.10	5.85	501	950	0.99
Lead hydroxide	Pb(OH) <sub>2</sub>	White, amorphous	241.20	--	145	--	0.015 5
Lead nitrate	Pb(NO <sub>3</sub> ) <sub>2</sub>	Colorless, cubic or monoclinical	331.2	4.53	Decomposes at 470	--	37.65
Lead oxide	PbO, litharge	Yellow, tetrahedral	223.19	9.53	886	--	0.001 7
Lead sulfate	PbSO <sub>4</sub> , Natural anglesite	White, monoclinical or rhombic	303.25	6.2	1170	--	0.004 25
Lead sulfide	PbS, Natural galena	Bluish metallic, cubic	239.25	7.5	1114	--	0.000 086

Source: CRC Handbook of Chemistry and Physics 1994.

**Table 2. Anthropogenic Lead Emissions into the Canadian Environment**

Source	Annual releases of lead (t)		
	Air	Land	Water
Industrial processes			
Mining and milling	964	38 000	49
Smelting and refining	2 024	22 000	29
Primary iron and steel	600	-	9
production	228	-	<1
Foundries	89	-	2
Metal fabrication	-	-	5
Metal plating	2	-	-
Metallurgical coke production	6	-	-
Cement manufacture	10	-	-
Leaded glass production	43	131	37
Chemical manufacturing	20	-	-
Tar sands operations	20	-	4
Battery storage operations	-	-	9
Pulp and paper mills	-	80	2
Refineries			
Fuel combustion			
Power generation	14	-	-
Industrial fuel combustion	8	-	-
Commercial fuel combustion	2	-	-
Residential fuel combustion	1	-	-
Gasoline-powered motor	7 000	-	-
vehicles	173	-	-
Aircraft			
Wastes			
Municipal sewage	-	-	412
Municipal refuse incineration	67	-	-
Sewage sludge incineration	2	-	-
Sewage sludge spreading	-	15	-
Waste oil incineration	130	-	-
Miscellaneous			
Gasoline marketing	63	-	-
Lead in ammunition	-	1 500	-
Miscellaneous	-	-	4
<b>TOTAL</b>	<b>11 466</b>	<b>61 726</b>	<b>562</b>

Source: Jaques 1985.

**Table 3. Existing Soil Criteria, Guidelines and Standards for Lead from Various Jurisdictions**

Jurisdiction	Category	Criterion/ guideline (mg·kg <sup>-1</sup> )	Reference
Canada	Interim Assessment Criterion: Interim Remediation criteria Agricultural Residential/parkland Commercial/industrial	25  375 500 1000	CCME 1991
Ontario	Remediation Criteria a) Surface soil (pH 5-9) Potable and non potable groundwater situations: Agricultural Land Use Residential/Parkland Industrial/Commercial  b) Sub-Surface Soil (pH 5-11) Potable and non-potable groundwater situations: Residential/Parkland Industrial/Commercial	  60 200 1000   1000 No value	OMOE 1994
Alberta	Tier 1 soil criterion for contaminated soil: assessment and remediation	50	Alberta Environmental Protection 1994
British Columbia	Clean-up Criteria: Background Requires site-specific evaluation Requires Immediate cleanup	50 500 1000	B.C. MOE 1989
Québec	Background Requires site-specific evaluation Immediate cleanup	50 500 1000	MEF 1994
New Jersey	Site-specific evaluation: Clean-up	250 1000	N.J. DEP 1990
Massachusetts	Soil standards S-1/GW-1, S-1/GW-2, S-1/GW-3 <sup>a</sup> S-2/GW-1, S-2/GW-2, S-2/GW-3 <sup>b</sup> S-3/GW-1, S-3/GW-2, S-3/GW-3 <sup>c</sup>	300 600 600	MDEP 1994
France	Site-specific evaluation	200	CCME 1991
United Kingdom	Remediation criteria for domestic gardens and allotments Remediation criteria for parks and open spaces	500 2000	CCME 1991
The Netherlands	Target value Requires intervention	85 530	RIVM 1994

- a Concentration based upon sensitive uses of the property and accessible soil (0-3 feet, unpaved), either currently or in the foreseeable future. Addition criteria are established for the protection of groundwater (GW-1, groundwater used as drinking water, GW-2, potential for volatile materials to migrate into indoor air, GW-3, environmental considerations for protection of surface water), based upon leaching from soils.
- b Concentration based upon property uses associated with moderate exposure ( children present) and accessible soils, either currently or in the foreseeable future. Addition criteria are established for the protection of groundwater (GW-1, groundwater used as drinking water, GW-2, potential for volatile materials to migrate into indoor air, GW-3, environmental considerations for protection of surface water), based upon leaching from soils.
- c concentrations based upon restricted access to property with limited potential for exposure, either currently or in the foreseeable future. Addition criteria are established for the protection of groundwater (GW-1, groundwater used as drinking water, GW-2, potential for volatile materials to migrate into indoor air, GW-3, environmental considerations for protection of surface water), based upon leaching from soils.

**Table 4. Toxicity Studies on Microbial Processes Selected for Use in Deriving Soil Quality Guidelines for Lead**

Microbial process (length of exposure)	Effect	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
Nitrification (6 weeks)	11% reduction	EC†	1000	PbO	7.0	Sandy soil; clay 5.5%, silt 12%, sand 82.5%; OC 2.2 %	Nominal	Bhuiya and Cornfield 1974
	9% reduction	EC†	1000	PbO	7.7			
Nitrification (10 d)	26% reduction	EC†	1625	Pb acetate	5.8	Silt 39%, clay 23%, sand 38%; OC 2.6%	Nominal	Liang and Tabatabai 1978
	7% reduction	EC†	1625	Pb acetate	7.4	Silt 50%, clay 34%, sand 16%; OC 5.5%		
	10% reduction	EC†	1036	Pb acetate	7.8	Silt 44%, clay 30%, sand 26%; OC 3.7%		
Nitrogen mineralization (6 weeks)	27% reduction	EC†	1000	PbO	7.7	Sandy soil; clay 5.5%, silt 12%, sand 82.5%; OC 2.2 %	Nominal	Bhuiya and Cornfield 1974
Nitrogen mineralization (20 d)	17% reduction	EC†	1036	Pb acetate	5.8	Silt 39%, clay 23%; sand 38%; OC 2.6%	Nominal	Liang and Tabatabai 1977
	19% reduction	EC†	1036	Pb acetate	6.6	Silt 54%, clay 45%; sand 1%; OC 3%		
	10% reduction	EC†	1036	Pb acetate	7.4	Silt 50%, clay 34%; sand 16%; OC 5.5%		
	28% reduction	EC†	1036	Pb acetate	7.8	Silt 44%, clay 30%; sand 24%; OC 3.7%		
Nitrogen mineralization (4 weeks)	32% reduction 44% reduction	EC EC	1000 4000	PbCl <sub>2</sub>	6.0	Sandy cambisol; clay 9%, silt 12%, sand 79%, OC 1.2%	Nominal	Wilke 1989
Denitrificaton (3 weeks)	No effect 15% reduction	-- EC	500 1000	Pb(NO <sub>3</sub> ) <sub>2</sub>	6.8	Silt loam; sand 8.5%, silt 63.4%, clay 28.1%; OC 1.8%	Nominal	Bollag and Barabasz 1979
Respiration - CO <sub>2</sub> release (8 weeks)	6% reduction 25% reduction	EC EC	10 100	Pb(SO <sub>3</sub> ) <sub>2</sub>	4.9	Loamy sand; sand 82%, silt 9.9%, clay 5.2%; OC 2.1%	Nominal	Cornfield 1977

Microbial process (length of exposure)	Effect	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
Respiration - CO <sub>2</sub> release (70 weeks)	9% increase 12% reduction 4% reduction 15% reduction 59% reduction	EC EC EC EC EC	150 400 1000 3000 8000	PbCl <sub>2</sub>	7.0	Sandy soil; clay 2%, silt 5%, sand 93%; OM 1.6%	Nominal	Doelman and Haanstra 1984
Respiration - CO <sub>2</sub> release (43 weeks)	6% reduction 23% reduction 45% reduction	EC EC EC	150 400 1000	PbCl <sub>2</sub>	6.0	Sandy loam; clay 9%, silt 26%, sand 65%; OM 5.7%		
Respiration - CO <sub>2</sub> release (90 weeks)	12% increase 7% reduction 8% reduction 10% increase 7% reduction	EC EC EC EC EC	150 400 1000 3000 8000	PbCl <sub>2</sub>	7.7	Silty loam; clay 19%, silt 74%, sand 7%; OM 2.4%		
Respiration - CO <sub>2</sub> release (12 weeks)	11% reduction	EC†	1000	PbO	6.0	Sandy soil; clay 5.5%, silt 12%, sand 82.5%; OC 2.2 %	Nominal	Bhuiya and Cornfield 1972
Respiration - O <sub>2</sub> consumption (42 h)	17% reduction 35% reduction 45% reduction 67% reduction	EC EC EC EC	375 750 1500 3000	PbCl <sub>2</sub>	5.7	Sandy soil; clay 12%, silt 81%, sand 7%; OM 2.8%	Nominal	Doelman and Haanstra 1979
	16% reduction 11% reduction 15% reduction 84% reduction	EC EC EC EC	375 750 1500 3000	PbCl <sub>2</sub>	5.4	Sandy soil; clay 28%, silt 66%, sand 6%; OM 6.7%		
	No effect 7% reduction 13% reduction 35% reduction	-- EC EC EC	375 750 1500 3000	PbCl <sub>2</sub>	7.5	Clay soil; clay 99%, sand 1%; OM 3.2%		

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

†Single concentration study.

**Table 5. Invertebrate and Plant Toxicity Studies Selected for Use in Deriving Soil Quality Guidelines for Lead**

Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
<b>Invertebrates</b>								
Earthworm ( <i>Eisenia fetida</i> )	Mortality (14 and 56 d)  Cocoon production (56 d)	NOEC (56 d) LC <sub>50</sub> (14 d) LC <sub>50</sub> (56 d)  NOEC EC <sub>50</sub>	2 190 4 480 3 760  1 810 1 940	Pb(NO <sub>3</sub> ) <sub>2</sub>	6.3	Sandy clay, OM 10%	Nominal	Spurgeon et al. 1994
Earthworm ( <i>Lumbricus rubellus</i> )	Growth (12 weeks)	NOEC LOEC (41% reduction)	1 000 3 000	PbCl <sub>2</sub>	7.3	Sandy loam	HCl with AAS	Ma 1982
Earthworm ( <i>Dendrobaena rubida</i> )	Total no. hatchlings (4 months)	EC (11% reduction) EC (22% reduction)	100 100	Pb(NO <sub>3</sub> ) <sub>2</sub>	4.5 5.5	Sandy soil and cattle dung	Nominal	Bengtsson et al. 1986
Earthworm ( <i>Eisenia fetida</i> )	Mortality (14 d)	LC <sub>50</sub>	5 941	Pb(NO <sub>3</sub> ) <sub>2</sub>	6.0	EEC artificial soil	Nominal	Neuhauser et al. 1985
Earthworm ( <i>Eisenia fetida</i> )	Mortality (14 d)	NOEC LC <sub>25</sub> LC <sub>50</sub> LC <sub>70</sub>	1 480 2 067 2 500 3 070	PbCl <sub>2</sub>	4.3	Artificial soil, OM 7.1%	HNO <sub>3</sub> , H <sub>2</sub> O <sub>2</sub> , and HCl, with ICP	Environment Canada 1995
<b>Plants</b>								
Red oak ( <i>Quercus rubra</i> )	Total dry weight (16 weeks)	NOEC LOEC (26% reduction)	20 50	PbCl <sub>2</sub>	6.0	Sandy loam, OM 1.5%	Nominal	Dixon 1988



Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
Loblolly pine ( <i>Pinus taeda</i> )	Height (19 weeks)	NOEC LOEC (30% reduction)	556.5 1 179	PbCl <sub>2</sub>	5.8	Fine loam	0.2 M HCl with AAS	Seiler and Paganelli 1987
	Dry shoot weight (19 weeks)	NOEC LOEC (38% reduction)	556.5 1 179					
	Dry root weight (19 weeks)	NOEC LOEC (61% reduction)	556.5 1 179					
Oats ( <i>Avena sativa</i> )	Seed yield (12 weeks)	NOEC†	250	Pb(NO <sub>3</sub> ) <sub>2</sub>	4.5, 6.4	Fine sandy loam	Nominal	Allinson and Dzialo 1981
Ryegrass ( <i>Lolium hybridum</i> )	Seed yield (12 weeks)	NOEC†	250	Pb(NO <sub>3</sub> ) <sub>2</sub>	4.5, 6.4	Fine sandy loam	Nominal	Allinson and Dzialo 1981
Oats ( <i>Avena sativa</i> )	Root biomass (6 weeks)	NOEC LOEC (37% reduction)	100	PbCl <sub>2</sub>	5.4	Loamy sand	Nominal	Khan and Frankland 1984
			500					
Wheat ( <i>Triticum aestiva</i> )	Root biomass (6 weeks)	NOEC LOEC (15% reduction)	100 500	PbCl <sub>2</sub>	5.4	Loamy sand	Nominal	Khan and Frankland 1984
Tomato ( <i>Lycopersicum esculentum</i> )	Dry fruit weight (at flowering)	NOEC LOEC (25% reduction)	300 600	Pb(NO <sub>3</sub> ) <sub>2</sub>	8.8	Fine sandy loam, OM 0.4%	Nominal	Khan and Khan 1983
Eggplant ( <i>Solanum melongena</i> )	Dry shoot weight (at flowering)	NOEC LOEC (42% reduction)	300 600	Pb(NO <sub>3</sub> ) <sub>2</sub>	8.8	Fine sandy loam, OM 0.4%	Nominal	Khan and Khan 1983
Corn ( <i>Zea mays</i> )	Root elongation (5 d)	NOEC LOEC (19% reduction)	100 250	PbCl <sub>2</sub>	6.5	Loamy sand	Nominal	Hassett et al. 1976
Corn ( <i>Zea mays</i> )	Dry shoot weight (17, 24, and 31 d)	LOEC (17 d) (18% reduction)	125	PbCl <sub>2</sub>	6.0	Loamy sand	Nominal	Miller et al. 1977
		LOEC (24 d) (29% reduction)	125					
		LOEC (31 d) (13% reduction)	125					

Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
Radish ( <i>Raphanus sativa</i> )	Seedling emergence (14 d)	NOEC EC <sub>25</sub> EC <sub>50</sub> LOEC (29% reduction)	421 833 1 236 974	PbCl <sub>2</sub>	4.1	Artificial soil, OM 6.3%	HNO <sub>3</sub> , H <sub>2</sub> O <sub>2</sub> , and HCl, with ICP	Environment Canada 1995
Lettuce ( <i>Lactuca sativa</i> )	Seedling emergence (14 d)	NOEC EC <sub>25</sub> EC <sub>50</sub> LOEC (46% reduction)	416 667 876 740	PbCl <sub>2</sub>	4.1	Artificial soil, OM 6.3%	HNO <sub>3</sub> , H <sub>2</sub> O <sub>2</sub> , and HCl, with ICP	Environment Canada 1995
Tomato ( <i>Lycopersicum esculentum</i> )	Dry weight (12 weeks)	EC (2% reduction) EC (18% reduction) EC (25% reduction) EC (39% reduction)	25 50 100 200	Pb(NO <sub>3</sub> ) <sub>2</sub>	8.2	Sandy soil, OM 0.9%	Nominal	Balba et al. 1991
	Dry weight (12 weeks)	EC (3% increase) EC (21% reduction) EC (35% reduction) EC (40% reduction)	25 50 100 200	Pb(NO <sub>3</sub> ) <sub>2</sub>	7.5	Clay soil, OM 2.1%		
Radish ( <i>Raphanus sativa</i> )	Shoot growth reduction (6 weeks)	EC <sub>50</sub>	2 800	PbCl <sub>2</sub>	5.4	Loamy sand	Nominal	Khan and Frankland 1983
	Root growth reduction (6 weeks)	EC <sub>50</sub>	2 400					
	Shoot growth reduction (6 weeks)	EC <sub>50</sub>	12 000	PbO				
	Root growth reduction (6 weeks)	EC <sub>50</sub>	10 000					
Onion ( <i>Allium cepa</i> )	Dry weight (8 weeks)	LOEC (11% reduction)	50	Pb(NO <sub>3</sub> ) <sub>2</sub>	8.3	Sand 40%, silt 35%, clay 24%	Nominal	Dang et al. 1990
Fenugreek ( <i>Trigonella poenumgraceum</i> )	Dry weight (8 weeks)	NOEC LOEC (20% reduction)	200 400	Pb(NO <sub>3</sub> ) <sub>2</sub>	8.3	Sand 40%, silt 35%, clay 24%	Nominal	Dang et al. 1990

Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
Wheat ( <i>Triticum estiva</i> )	Dry stem weight (15 weeks)	EC (15% reduction)	300	PbO	5.9	Alluvial soil	Nominal	Muramoto et al. 1990
		EC (7% reduction)	1 000					
		EC (24% reduction)	3 000					
		EC (31% reduction)	10 000					
	Dry root weight (15 weeks)	EC (13% increase)	300					
		EC (12% reduction)	1 000					
		EC (37% reduction)	3 000					
		EC (44% reduction)	10 000					
	Dry grain weight (15 weeks)	No effect	300					
		EC (3% reduction)	1 000					
		No effect	3 000					
		EC (29% reduction)	10 000					

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

†Single concentration study.

Table 6. Additional Invertebrate and Plant Toxicity Studies

Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference						
Invertebrates														
Woodlice ( <i>Porcellio scaber</i> )	Young produced (64 weeks)	EC (5% reduction)	1 600	PbO	4.6	Soil litter	Nominal	Beyer and Anderson 1985						
		EC (10% reduction)	3 200											
		EC (22% increase)	6 400											
		EC (68% reduction)	12 800											
	Adult survival (64 weeks)	EC (2% reduction)	1600											
		EC (14% reduction)	3 200											
		EC (5% reduction)	6 400											
		EC (27% reduction)	12 800											
	Young survival (64 weeks)	NOEC	6 400											
		LOEC (25% reduction)	12 800											
	Earthworm ( <i>Eisenia fetida</i> )	Weight (8 weeks)	NOEC LOEC (62% reduction)						10 000 12 000	Pb acetate	NR†	Horse manure and soil	Nominal	Malecki et al. 1982
Cocoon production (20 weeks)		NOEC LOEC (50% reduction)	2 000 4 000											
Plants														
Red spruce ( <i>Picea rubens</i> )	Height (19 weeks)	LOEC (38% reduction)	150.1	PbCl <sub>2</sub>	3.74	Mountain spodosol	0.2 M HCl with AAS	Seiler and Paganelli 1987						
	Dry shoot weight (19 weeks)	LOEC (45% reduction)	150.1											
	Dry root weight (19 weeks)	LOEC (43% reduction)	150.1											
	Photosynthesis (19 weeks)	NOEC LOEC (30% reduction)	150.1 271.1											

Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
American sycamore ( <i>Plantanus occidentalis</i> )	Photosynthesis (90 d)	EC (25% reduction)	100	PbCl <sub>2</sub>	NR	Silty clay loam	Nominal	Carlson and Bazzaz 1977
		EC (22% reduction)	250					
		EC (40% reduction)	500					
		EC (46% reduction)	1 000					
	Foliage biomass (90 d)	EC (30% reduction)	50					
		EC (22% reduction)	100					
		EC (30% reduction)	250					
		EC (22% reduction)	500					
		EC (35% reduction)	1 000					
Autumn olive ( <i>Elaeagnus umbellata</i> )	Photosynthesis (7 weeks)	EC (3% reduction)	20	PbCl <sub>2</sub>	NR	Sandy peat	Nominal	Rolfe and Bazzaz 1975
		EC (5% reduction)	40					
		EC (7% reduction)	80					
		EC (5% reduction)	160					
		EC (18% reduction)	320					
Loblolly pine ( <i>Pinus taeda</i> )	Photosynthesis (7 weeks)	No effect	20	PbCl <sub>2</sub>	NR	Sandy peat	Nominal	Rolfe and Bazzaz 1975
		No effect	40					
		No effect	80					
		EC (4% reduction)	160					
		EC (12% reduction)	320					
Lettuce ( <i>Lactuca sativa</i> )	Leaf biomass (3 weeks)	EC‡ (36% reduction)	1 000	PbCl <sub>2</sub>	3.8	Silty clay loam	Nominal	John and van Laerhoven 1972
Oats ( <i>Avena sativa</i> )	Root biomass (3 weeks)	NOEC‡	1 000	PbCl <sub>2</sub>	3.8	Silty clay loam	Nominal	John and van Laerhoven 1972
	Top biomass (3 weeks)	NOEC‡	1 000					

Organism	Effect (length of exposure)	Endpoint*	Concentration (mg Pb·kg <sup>-1</sup> )	Form of lead	Soil pH	Test substrate	Extraction method	Reference
Sitka spruce ( <i>Picea sitchensis</i> )	Root yield (100 d)	EC (30% reduction)	5	PbCl <sub>2</sub>	3.3- 3.9	Peaty gley	Nominal	Burton et al. 1984
		EC (56% reduction)	10					
		EC (8% reduction)	25					
		EC (4% reduction)	50					
		EC (27% reduction)	100					
	Root length (100 d)	LOEC (41% reduction)	5					
	Shoot yield (100 d)	EC (36% reduction)	5					
		EC (34% reduction)	10					
		EC (12% reduction)	25					
		EC (4%r reduction)	50					
		EC (24% reduction)	100					

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

†NR = not reported.

‡Single concentration study.

**Table 7. Mammalian and Avian Oral Toxicity Studies Selected for Use in Deriving Soil Quality Guidelines for Lead**

Organism	Effect (length of exposure)	Endpoint*	Feed Concentration (mg Pb·kg <sup>-1</sup> )	Average feed intake (kg·d <sup>-1</sup> )	Average body weight (kg)	Average daily dose (mg Pb·kg <sup>-1</sup> bw·d <sup>-1</sup> )	Form of lead	Reference
Holstein steer calves	Body weight (12 weeks)	EC (6% reduction)	NR†	NR	61	3.9	PbCO <sub>3</sub>	Lynch et al. 1976
		EC (13% reduction)	NR	NR	61	7.7		
American kestrel hatchlings ( <i>Falco sparverius</i> )	Mortality (10 d)	EC (0% mortality)	NR	NR	0.068	125	Pb powder	Hoffman et al. 1985
		EC (40% mortality)	NR	NR	0.068	625		
American kestrel hatchlings ( <i>Falco sparverius</i> )	Body weight (10 d)	NOEC	NR	NR	0.068	25	Pb powder	Hoffman et al. 1985
		LOEC (16% reduction)	NR	NR	0.068	125		
Japanese quail ( <i>Coturnix coturnix japonica</i> )	Body weight (5 weeks)	EC (11% reduction)	500	0.0029‡	0.077	18.8‡	Pb acetate	Morgan et al. 1975
		EC (16% reduction)	1000	0.0029‡	0.077	37.7‡		
Japanese quail ( <i>Coturnix coturnix japonica</i> )	Body weight (12 weeks)	NOEC	100	0.0035‡	0.093	3.8‡	Pb acetate	Edens et al. 1976
		LOEC (21% reduction)	1000	0.0035‡	0.093	37.6‡		
Japanese quail ( <i>Coturnix coturnix japonica</i> )	Hatching of setable eggs (12 weeks)	NOEC	10	0.0035‡	0.093	0.4‡	Pb acetate	Edens et al. 1976
		LOEC (28% reduction)	100	0.0035‡	0.093	3.8‡		
Japanese quail ( <i>Coturnix coturnix japonica</i> )	Egg production (5 weeks)	NOEC	1	0.027	0.152	0.2‡	Pb acetate	Edens and Garlich 1983
		LOEC (28% reduction)	10	0.027	0.152	1.8‡		
Leghorn hens	Body weight (4 weeks)	NOEC	100	0.101	1.55	6.5‡	Pb acetate	Edens and Garlich 1983
		LOEC (7% reduction)	200	0.101	1.55	13.0‡		
Leghorn hens	Egg production (4 weeks)	NOEC	25	0.101	1.55	1.7‡	Pb acetate	Edens and Garlich 1983
		LOEC (12% reduction)	50	0.101	1.55	3.3‡		
Wistar rat	Body weight (9 weeks)	NOEC LOEC (29% reduction)	NR NR	NR NR	NR NR	33 330	Pb acetate	Barratt et al. 1989

Organism	Effect (length of exposure)	Endpoint*	Feed Concentration (mg Pb·kg <sup>-1</sup> )	Average feed intake (kg·d <sup>-1</sup> )	Average body weight (kg)	Average daily dose (mg Pb·kg <sup>-1</sup> ·bw·d <sup>-1</sup> )	Form of lead	Reference
Japanese quail ( <i>Coturnix coturnix japonica</i> )	Weight reduction (5 weeks)	NOEC	100	0.027	0.152	17.7‡	Pb acetate	Edens and Garlich 1983
Sprague-Dawley rat	Weight reduction (30 d)	NOEC	250	NR	NR	25.7	Pb acetate	Freeman et al. 1992
American kestrel ( <i>Falco sparverius</i> )	Fertility and survival (7 months)	NOEC	50	0.00496‡	0.131	1.9‡	Pb powder	Pattee 1984
Ringed turtle doves ( <i>Streptopella risoria</i> )	Egg production§ (11 weeks)	NOEC	1000 mg·L <sup>-1</sup>	0.0034 L·d <sup>-1</sup> ‡	0.087	39.1‡	Pb acetate	Kendall and Scanlon 1981
Pig	Mortality (13 weeks)	NOEC	NR	NR	20.5	30.2	Pb acetate	Lassen and Buck 1979
Wether lamb	Weight reduction (12 weeks)	NOEC	1000	1.6‡	36	44.4‡	Pb acetate	Fick et al. 1976

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

†NR = not reported.

‡Calculated from standard equations for daily feed and water intake by livestock and wildlife (Warrington 1995, British Columbia Ministry of the Environment, Lands and Parks, pers. com.).

§Single concentration study.



**Table 8. Environmental Guidelines for Lead**

Guideline	Land use		
	Agricultural (mg Pb·kg <sup>-1</sup> )	Residential/ parkland (mg Pb·kg <sup>-1</sup> )	Commercial/ industrial (mg Pb·kg <sup>-1</sup> )
TEC or ECL*			
20th percentile	250	250	NA†
25th percentile	250	250	400
30th percentile	400	400	600
35th percentile	NA	NA	667
Nutrient and energy cycling check	643	643	787
SQG <sub>sc</sub>	250	250	400
SQG <sub>i</sub>	70	NA	NA
SQG <sub>e</sub>	70	250	400
CCME interim criteria§	375	500	1000

\*As per the CCME 1996 protocol, the TEC for agricultural and residential/parkland land uses corresponds to the 25th percentile of the no effects and effects distribution when using the weight of evidence method, while the ECL for commercial/industrial land use corresponds to the 25th percentile of the effects data distribution. The other percentiles are presented for comparison purposes only.

†NA = not applicable.

‡OMOE 1994.

§CCME 1991.

CANADIAN SOIL QUALITY  
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DATE	ISSUED TO

DATE:  DAT. No. 28-115 PRINTED IN CANADA