Ecosystem Health

Science-Based Solutions

Canadian Soil Quality Guidelines for Polychlorinated Biphenyls (PCBs): Environmental Health

Report No. 1-2

Environment Canada Environnement Canada

Canada

Science-Based Solutions
Solutions fondées sur la science

Prepared and published by
National Guidelines and Standards Office
Environmental Quality Branch
Environment Canada
Ottawa

August, 2001

ISSN 1497-2689
ISBN 0-662-30537-X
Cat. no. En40-236/1-2001E

Scientific Supporting Document

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NOTE TO READERS

The *Ecosystem Health: Science-based Solutions* series is dedicated to the dissemination of scientific knowledge, information and tools for monitoring, assessing, and reporting on ecosystem health to support Canadians in making sound decisions. Documents published in this series include the scientific basis, methods, approaches and frameworks for environmental guidelines and their implementation; monitoring, assessing, and rehabilitating environmental quality in Canada; and, indicator development, environmental reporting and data management. Issues in this series are published *ad libitum*.

This particular issue provides the scientific supporting information and rationale for the development of Canadian Soil Quality Guidelines for the protection of environmental health for polychlorinated biphenyls (PCBs). The information in this document is current as of 1999, when the document was originally prepared. Minor revisions and editorial changes have been made for publication in 2001. For additional information regarding these guidelines, please contact:

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Canadian Soil Quality Guidelines are approved by the Soil Quality Guidelines Task Group (formerly the Subcommittee on Environmental Quality Criteria for Contaminated Sites) of the Canadian Council of Ministers of the Environment (CCME). Environment Canada acts as the federal representative and serves as the technical secretariat to this Task Group. These guidelines are included in the *Canadian Environmental Quality Guidelines*, which was published by the CCME in October of 1999. For CCME publications, please contact:

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This scientific supporting document is available in English only. A factsheet is also available in English under the title *Canadian Environmental Quality Guidelines* (CCME 1999). Ce document scientifique du soutien n'est disponible qu'en anglais avec un résumé en français. Un feuillet d'information est aussi disponible en français sous le titre *Recommandations canadiennes pour la qualité de l'environnement* (CCME 1999).

Reference listing:
Environment Canada. 2001. Canadian Soil Quality Guidelines for Polychlorinated Biphenyls (PCBs): Environmental Health. Scientific Supporting Document. Ecosystem Health: Science-based Solutions Report No. 1-2. National Guidelines and Standards Office, Environmental Quality Branch, Environment Canada. Ottawa.

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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 scientific supporting document was prepared by the National Guidelines and Standards Office of the 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 1996a).

This document contains a review of information on the chemical and physical properties of PCBs, a review of sources and emissions in Canada, the distribution and behaviour of PCBs in the environment, and the toxicological effects of PCBs on microbial processes, plants, and animals. This information is used to derive soil quality guidelines for PCBs to protect ecological receptors in four types of land uses: agricultural, residential/parkland, commercial, and industrial. A new approach to deriving soil guidelines for persistent substances, such as PCBs, that are subject to long-range transport and have a strong tendency to bioaccumulate and biomagnify in food chains, is also presented in chapter 5. This approach is a modification of the existing soil and food ingestion guideline procedure intended to protect secondary and tertiary consumers from PCB contamination through terrestrial food chains.

Environmental soil quality guidelines have been derived for PCBs for each of the four land uses. The environmental soil quality guideline (SQG_E) relative to total PCBs for agricultural and residential/parkland land uses is 1.3 mg PCB· kg^{-1} soil, based on the soil contact guideline. The SQG_E for commercial and industrial land uses is 33 mg PCB· kg^{-1} soil, based on the soil contact guideline.

The Canadian Soil Quality Guidelines for the protection of environmental and human health, as recommended by the Canadian Council of Ministers of the Environment, and published in October of 1999 in *Canadian Environmental Quality Guidelines* (CCME 1999), for total PCBs are: 0.5 mg· kg^{-1} soil for agricultural land use, 1.3 mg· kg^{-1} soil for residential/parkland land use, 33 mg· kg^{-1} soil for commercial land use, and 33 mg· kg^{-1} soil for industrial land use. The guideline for agricultural land use is based on the 1991 interim soil quality criterion (CCME 1991) because it is lower than the environmental soil quality guideline derived in this document. The guidelines for the other three land uses are based on the environmental soil quality guidelines derived in this document. Human health soil quality guidelines for PCBs have not been developed, due to insufficient data.

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 scientifique justificatif a été préparé par le Bureau national des recommandations et des normes de la Direction de la qualité de l'environnement (Environnement Canada), qui sert de secrétariat technique pour le Groupe de travail du CCME sur les Recommandations 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 1996a).

Le document présente un survol des données sur les propriétés chimiques et physiques des BPC, les sources et les émissions de ces produits au Canada, leur distribution et leur comportement dans l'environnement, ainsi que leurs effets toxicologiques sur les processus microbiens, les végétaux et les animaux. Ces informations servent à élaborer des recommandations pour la qualité des sols concernant les BPC en vue de la protection des récepteurs écologiques pour quatre types d'utilisation des sols: agricole, résidentielle/parc, commerciale et industrielle. Le chapitre 5 présente aussi une approche nouvelle pour l'élaboration de recommandations pour les substances telles les BPC, qui sont persistantes, susceptibles d'être transportées sur de longues distances et qui ont tendance à être bioaccumulées et bioamplifiées dans les chaînes trophiques. Cette approche est une modification de la procédure pour l'élaboration de la recommandation relative à l'ingestion de sol et de nourriture afin de protéger les consommateurs secondaires et tertiaires de la contamination venant des chaînes alimentaires terrestres.

Des recommandations pour la qualité des sols concernant les BPC en vue de la protection de l'environnement ont été élaborées pour chacun des quatre types d'utilisation des sols. Au niveau des terrains à vocation agricole et résidentielle/parc, la recommandation pour la qualité des sols concernant les BPC en vue de la protection de l 'environnement (RQSE) est de 1,3 mg de BPC· kg^{-1} de sol. Pour les terrains à vocation commerciale et industrielle, elle est de 33 mg de BPC· kg^{-1} de sol.

Les Recommandations canadiennes pour la qualité des sols en vue de la protection de l'environnement et de la santé humaine, préconisées par le Conseil canadien des ministres de l'environnement et publiées en octobre 1999 dans le document intitulé *Recommandations canadiennes pour la qualité de l'environnement* (CCME, 1999) concernant les BPC totaux sont les suivantes : 0,5 mg· kg^{-1} de sol pour les terrains à

vocation commerciale, 1,3 mg· kg^{-1} de sol pour les terrains à vocation résidentielle/parc, 33 mg· kg^{-1} de sol pour les terrains à vocation commerciale et 33 mg· kg^{-1} de sol pour les terrains à vocation industrielle. La recommandation pour les terrains à vocation agricole est fondée sur le critère provisoire de qualité des sols fixé en 1991 (CCME, 1991) parce qu'il est inférieur à la recommandation pour la qualité des sols pour la protection de l'environnement établie dans le présent document. Les recommandations visant les trois autres types d'utilisation sont fondées sur les recommandations pour la qualité des sols (environnement) qui se trouvent dans le présent document. Les recommandations pour la qualité des sols en vue de la protection de la santé humaine n'ont pas été élaborées en raison du manque de données.

CHAPTER 1. INTRODUCTION

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

The development of Canadian Soil Quality Guidelines was initiated through the National Contaminated Sites Remediation Program (NCSRP) in 1991, by the CCME Subcommittee on Environmental Quality Criteria for Contaminated Sites. In response to the urgent need to begin remediation of high priority "orphan" contaminated sites, an interim set of soil quality criteria was adopted from values that were in use in various jurisdictions across Canada (CCME 1991). Although the NCSRP program officially ended in March of 1995, the development of soil quality guidelines was pursued under the direction of the CCME Soil Quality Guidelines Task Group because of the continued need for national soil quality guidelines for the management of soil quality (with a particular focus on remediation of contaminated sites). Environment Canada serves as the technical secretariat to this Task Group.

Canadian Soil Quality Guidelines are developed according to procedures that have been described by the CCME (CCME 1996, 1997, and reprinted in 1999). According to this protocol, both environmental and human health soil quality guidelines are developed for four land uses: agricultural, residential/parkland, commercial, and industrial. The lowest value generated by the two approaches for each of the four land uses is recommended by the CCME as the Canadian Soil Quality Guideline. Guidelines for a number of substances were developed using this protocol and released in a working document entitled *Recommended Canadian Soil Quality Guidelines* (CCME 1997). The guidelines originally published in that document have since been revised and are now superceded by the Canadian Soil Quality Guidelines for the protection of environmental and human health published by the CCME in October of 1999 (CCME 1999). The interim soil quality criteria (CCME 1991) should be used only when soil quality guidelines based on the CCME protocol have not yet been developed for a given chemical.

This scientific supporting document provides the background information and rationale for the derivation of environmental soil quality guidelines for PCBs. This document contains a review of information on the chemical and physical properties of PCBs, a review of sources and emissions in Canada, the distribution and behaviour of PCBs in the environment, and the toxicological effects of PCBs on terrestrial mammals, plants, and soil organisms. The CCME recognizes that persistent bioaccumulative substances should be virtually eliminated from the environment. Nevertheless, the CCME also recognizes the

need for remediation guidelines, as interim management objectives, for persistent bioaccumulative substances in soils. Thus, the reviewed information is used to derive guidelines for PCBs to protect ecological receptors according to the processes outlined in A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines (CCME 1996a) for agricultural, residential/parkland, commercial and industrial land uses.

The Canadian Soil Quality Guidelines presented in this document are intended as general guidance. Site-specific conditions should be considered in the application of these values. The reader is referred to CCME (1999) for further generic implementation guidance pertaining to the guidelines. Soil quality guidelines are derived to approximate a "no- to low-" effect level (or threshold level) based only on the toxicological information and other scientific data (fate, behaviour, etc.) available for the substance of concern, and they do not consider socioeconomic, technological, or political factors. These non-scientific factors are to be considered by site managers at the site-specific level as part of the risk management process. Because these guidelines may be used and applied differently across provincial and territorial jurisdictions, the reader should consult the laws and regulations of the jurisdiction they are working within for applicable implementation procedures.

CHAPTER 2. BACKGROUND INFORMATION

Physical/Chemical Properties

Polychlorinated biphenyls (PCBs) (CAS No. 1336-36-3) refer to a group of 209 compounds consisting of chlorine atoms substituted on a biphenyl ring. Varying degrees of chlorination of the biphenyl ring give rise to ten PCB congener groups, with a number of positional isomers possible within each group (Eisler 1986). The identities of individual PCB compounds are determined by the number and locations of chlorine atoms substituted on the biphenyl molecule (Figure 1) and are referred to by a numbering system established by the International Union of Pure and Applied Chemistry (IUPAC) (Appendix I). PCB congeners with chlorine atoms in non-ortho positions only (i.e., positions 3, 3', 4, 4', 5, 5') are defined as coplanar PCBs. Regulations under the Canadian Environmental Protection Act (CEPA) define PCBs as chlorobiphenyls that have the molecular formula $C_{12}H_{10-n}Cl_n$ where n is greater than two (i.e., tri- to deca-chlorobiphenyls) (CEPA 1999).

Prior to 1980, commercial PCB formulations were sold under a variety of trade names, including Aroclor, Askarel, Chloretol, Dyknol, Inerteem, Kanechlor, Noflamol, Phenoclor and Pyranol (Hutzinger et al. 1974; Eisler 1986). All PCBs manufactured in North America were produced by Monsanto Co. under the trade name Aroclor. Aroclor mixtures are designated by a four digit code, with the first two numbers representing the type of compound (12 = chlorinated biphenyl) and the last two numbers representing the percent chlorine by weight (Hutzinger et al. 1974; Kalmaz and Kalmaz 1979). Aroclor 1016 is an exception to this code and is a redistilled version of Aroclor 1242 with a chlorine composition of 41% (Safe 1994). The physical appearance and chlorination levels of commercial PCB formulations vary, with low chlorine mixtures appearing as clear mobile oils and high chlorine mixtures as sticky yellow resins and waxy white solids (Hutzinger et al. 1974; Pal et al. 1980; Addison 1986).

In general, PCBs are extremely stable compounds that degrade slowly under normal environmental conditions. PCBs are characterised by high thermal and chemical stabilities, low vapour pressures, high dielectric constants, high electric resistivities, high densities, hydrophobicity and high lipophilicity (Appendix II) (Hutzinger et al. 1974; Pal et al. 1980). These properties are accentuated with increasing chlorination (Pal et al. 1980; Mackay et al. 1992). Non-coplanar PCBs are slightly more reactive than coplanar PCBs (Strachan 1988). The melting points of PCBs range from 16°C to 305°C (Mackay et al. 1992) and boiling points are approximately 267°C (Pal et al. 1980). PCBs are extremely resistant to oxidation, reduction, addition, elimination, electrophilic substitution and hydrolysis (Pal et al. 1980; Moore and Walker 1991).

The same properties that make PCBs useful to industry also make them persistent and potentially hazardous in the environment. Once released to the environment, PCBs persist for years and possibly for decades (Addison 1986). The stable and lipophilic nature of PCBs indicates a high potential for adsorption in soils with organic content and for

bioaccumulation in the fatty tissues of animals (Addison 1986; Mackay et al. 1992), while their hydrophobicity, stability and ability to adhere to particulates indicate a potential for long-range atmospheric transport (Barrie et al. 1992). The complexity of and number of components within PCB mixtures present immense analytical and toxicological problems (Addison 1986). Toxicological problems are also aggravated by the fact that some of the trace impurities, such as polychlorinated dibenzofurans (PCDF), in the commercial PCB mixtures may be more toxicologically significant than the PCBs themselves (Addison 1986, Eisler 1986).

Analytical Methods

Environment Canada has developed a comprehensive procedure for analysing PCBs (Environment Canada 1997). This procedure is intended for use in support of the various regulations that govern the use and disposal of PCB-containing materials. The procedure is comprised of two parts: a fast and economical screening method that uses high resolution gas chromatography/electron capture detection to determine if PCBs are present in a sample and to provide an estimate of their concentration; and a confirmative method that uses gas chromatography/mass spectrometry to identify and quantify total PCBs. PCBs are quantified on a total homologue basis, as opposed to individual congeners or Aroclors.

Other analytical methods recommended for PCBs by the CCME are described in CCME (1993a). These methods provide guidance for sampling and analysing environmental matrices such that the data are representative, consistent and comparable. There are three analytical methods recommended for PCBs; these are summarised briefly below.

US EPA Method 505, Revision 2, is applicable to the analysis of drinking water and raw source water, including most surface water and groundwater sources. This is a very sensitive gas chromatographic/electron capture method that is more useful for monitoring than for exploratory analyses. Aroclors 1016, 1221, 1232, 1242, 1248, 1254 and 1260 are covered by this method; detection limits range from 0.08 to 15 $\mu g \cdot L^{-1}$.

Standard Method 6410B is applicable to the determination of PCBs in municipal and industrial discharges that are partitioned into an organic solvent and are amenable to gas chromatography. This method is a broad-spectrum gas chromatographic/mass spectrometric method for the detection of semi-volatile compounds following liquid-liquid extraction. Analytes covered by this method include Aroclors 1016, 1221, 1232, 1242, 1248, 1254 and 1260. The detection limits for Aroclors 1221 and 1254 are 30 $\mu g \cdot L^{-1}$ and 36 $\mu g \cdot L^{-1}$, respectively; detection limits for other Aroclor mixtures are unclear.

US EPA Method 8080B, Revision 2, is used to determine the concentration of PCBs in extracts prepared from water, groundwater, soils and sediments. This gas chromatography/electron capture method is relatively inexpensive and suitable for monitoring analyses. Analytes covered by this method include Aroclors 1016, 1221, 1232,

1242, 1248, 1254 and 1260. Following corrections for a 30g soil sample digested, with final extract being brought to 10 ml, a detection limit of 2.17×10^{-5} mg PCB·kg^{-1} soil would be calculated for this method regarding Aroclor 1242. Detection limits for other Aroclor mixtures are not reported.

Production, Uses and Global Sources of PCBs

PCBs are commercially produced by chlorination of a biphenyl with anhydrous chlorine in the presence of iron fillings or ferric chloride as a catalyst (Hutzinger et al. 1974). The product of this process is a complex mixture of chlorobiphenyls containing 18 to 79% chlorine by weight, with the precise composition of the mixture depending upon the conditions under which chlorination occurred (Eisler 1986). Polychlorinated dibenzofurans (PCDF) may occur as post-production toxic impurities in PCBs at levels of 0.8 to 33 mg·kg^{-1} (Addison 1986; Eisler 1986). Heat action on and electrical arcing through PCBs used in electrical capacitors and transformers may result in the formation of very high levels (5000 to 12 000 mg·kg^{-1}) of PCDFs in the mixture (Eisler 1986).

PCBs were first prepared in 1881 and began to be used in industry in 1929 (Pal et al. 1980; CCREM 1986; Tanabe 1988). PCBs have many industrial applications. Their dielectric properties, chemical stability at high temperatures and low vapour pressures make them ideal for use as electric insulators, plasticisers for adhesives, lubricants, hydraulic fluids, liquid seals, cutting oils, heat transfer agents, flame retardants, antifouling agent in paints, ink solvents in carbonless copying paper, waterproofing materials and vacuum diffusion pump oils (Kalmaz and Kalmaz 1979; Pal et al. 1980; Barrie et al. 1992; Addison 1986; CCREM 1986). After 1971, PCBs were primarily used as insulating and cooling agents in closed electrical systems such as capacitors and transformers (Eisler 1986; Strachan 1988).

Roughly 1.2 million metric tonnes of PCBs were produced world-wide from 1929 to 1977 (Tanabe 1988; WHO 1993), with just over half that amount (635 000 t) produced by Monsanto Co. in the United States under the trade name Aroclor (CCREM 1986). An estimated 31% (370 000 t) of the total world production of PCBs is estimated to have been released into the global environment prior to restrictions on use, while 65% is still in use or in storage and 4% has been destroyed in waste incineration plants (Tanabe 1988). North American manufacturing and processing of PCBs was discontinued in 1979 as a result of environmental and health concerns (WHO 1993; Eisler 1986). Although PCBs were not manufactured in Canada, approximately 40 000 t were imported into the country and used commercially from 1929 to 1977 (CCREM 1986). Twenty-four thousand tonnes were reported to be in use (approximately 17 520 t) or in storage (approximately 6480 t) in 1986, with the remaining 16 000 t unaccounted for and assumed either to be in mineral oils at low levels of concentration or to have been released to the environment (CCREM 1986; Strachan 1988). The use of PCBs in Canada has been steadily declining as PCB-containing equipment is removed from service. In 1993, 11 505 t (net weight) of

PCBs were reported to be in use in transformers, capacitors and other equipment, while 133 291 t (gross weight) of PCB-contaminated wastes were in storage (CCME 1993b).

Legislation limiting the importation and severely restricting the use of PCBs in Canada was first introduced in 1977 and importation was prohibited in 1980 (Strachan 1988; Barrie et al. 1992). By 1985, PCBs were limited to use as a dielectric fluid in existing equipment (Strachan 1988). Under current legislation, PCB-containing electrical equipment manufactured prior to 1980 may remain in use until the service life of the equipment has ended (CCREM 1986). Strict maintenance and handling procedures, as well as regulatory control and monitoring by federal and provincial governments, are currently in place to reduce the risk of releases into the environment (CCREM 1986).

Sources and Levels in the Canadian Environment

PCBs are man-made synthetic compounds that generally do not occur naturally in the environment. PCBs were first discovered in the environment in 1966 (Jensen 1966 cited in Tanabe 1988). Since then, PCBs have become widespread in the global environment, with detectable concentrations occurring even in remote areas such as the Arctic and Antarctic atmosphere, hydrosphere and biosphere (Tanabe 1988). PCBs are ubiquitous in Canada and all organisms have been exposed to them. PCB contamination of the Canadian environment is greatest in the region of the Great Lakes, followed by Quebec and the Atlantic provinces and then British Columbia and the Prairies. The rank of the northern territories is unknown (Strachan 1988).

Several studies have reported that PCB residue levels in the environment are gradually declining (Addison et al. 1986; Tanabe 1988). In most cases, the downward trends were observed at sites where restrictions on use have resulted in decreased local emissions (e.g. Lake Ontario) (Tanabe 1988). However, substantial quantities of PCBs remain in use and PCBs are stored in landfills in large quantities; furthermore, PCBs are persistent and appear to be recycled by the environment (Murphy et al. 1985). Thus, PCB levels in the global environment, particularly in remote areas, are unlikely to decrease in the near future (Tanabe 1988).

Air

Releases of PCBs to the atmospheric environment have occurred through vaporisation of plasticisers, evaporation from paints and wood preservatives, evaporation from spilled and leaked PCBs from transformers, hydraulic systems and other PCB-containing equipment, evaporation from waste PCB fluids used to oil roads, incomplete incineration of PCB-contaminated waste; volatilisation from landfill sites, and emissions from engines and furnaces burning fuels containing PCBs (Buckley 1982; Murphy et al. 1985; Strachan 1988).

Landfills are a continuing source of PCBs to the atmospheric environment with U.S. landfills estimated to release 10 to 100 $kg \cdot a^{-1}$ of PCBs to the atmosphere (Murphy et al. 1985). Long-term measurements of atmospheric PCBs in the vicinity of landfills containing these substances were reported to be 1.7 to 3.8 $ng \cdot m^{-3}$ in summer and 0.3 to 0.6 $ng \cdot m^{-3}$ in winter (Hermanson and Hites 1989). Deposition of PCBs and PCB-containing materials into landfills was the primary method of disposal for these chemicals in North America until the mid-1970s (Strachan 1988). PCB-containing carbonless copy paper and small capacitors used in a multitude of consumer products continue to be routinely disposed of this way (Murphy et al. 1985). From 45% to 50% of the PCBs produced in North America is estimated to have been placed in landfills and dumps (Strachan 1988).

Landfills have occasionally been set on fire to reduce the volume of solid waste within them (Strachan 1988). PCBs are destroyed by combustion only at temperatures over 1100°C (Addison 1986; CCREM 1986). Low-temperature incineration of PCB-contaminated waste can disperse PCBs through vaporisation and in association with fly ash, and may also result in the formation and volatilisation of toxic PCDFs (Addison 1986; Strachan 1988). Anaerobic decomposition of organic wastes in sanitary landfills results in the production and emission of methane and carbon dioxide gases. PCBs and other volatile materials present in landfills may be transported into the atmosphere in association with the emissions of these gases (Murphy et al. 1985).

PCBs are widely present in the Canadian atmosphere and have been detected in precipitation in all parts of the country (Strachan 1988). In general, atmospheric PCB concentrations are highest over industrialised areas or landfills, and are present at lower levels over oceans and non-industrialised regions such as the northern territories (WHO 1993). Ambient air concentrations have been estimated at 1 to 2 $ng \cdot m^{-3}$ in the Great Lakes region, although concentrations as high as 6.7 $ng \cdot m^{-3}$ have been recorded (Strachan 1988). Mean total PCB concentrations of 0.015 $ng \cdot m^{-3}$ and 0.018 $ng \cdot m^{-3}$ were measured during the summers of 1986 and 1987, respectively, near Ellesmere Island in the high Arctic (Patton et al. 1989), while levels of 0.002 to 0.07 $ng \cdot m^{-3}$ have been recorded in the Northwest Territories (Bidleman et al. 1976).

Precipitation concentrations of PCBs indirectly reflect atmospheric levels (Strachan 1988). In 1984, rain from such widely separated places as New Brunswick, northern Saskatchewan and the middle of Lake Superior consistently showed PCB levels of at least 0.001 to 0.003 $\mu g \cdot L^{-1}$ (Strachan 1988). Mean annual PCB concentrations of 0.18 $\mu g \cdot L^{-1}$ and 0.17 $\mu g \cdot L^{-1}$ were measured in precipitation near Ellerslie, Prince Edward Island in 1982 and 1983, respectively. These elevated levels were attributed to an unknown local source. Observed precipitation PCB levels at the same site were an order of magnitude lower the following year (Brun 1983, 1984, 1985).

Water and Sediments

The major sources of PCBs to aquatic systems are leaks, spills, municipal effluents, runoff contaminated soils, leachates from unsecured landfills, and wet and dry deposition from the atmosphere (Strachan 1988; Moore and Walker 1991; WHO 1993). Release of PCBs in the effluents of manufacturing industries was an important historical mode of entry into the aquatic environment for these chemicals before their production and use was limited by legislation (NRC 1979). Current industrial processes of concern are the recycling of waste oil and paper (Strachan 1988). PCB-containing fluids are sometimes inadvertently combined with waste oil destined for recycling and may be released to effluents, while PCBs used in the manufacture of inks and carbonless copy papers are released to effluent streams or incorporated into sewage sludge following the de-inking and pulping processes of paper recycling (Strachan 1988).

Aquatic environments with high levels of PCB contamination tend to occur near heavily industrialised areas with active emission sources. Polluted rivers, lakes and estuaries have been shown to contain higher levels of PCBs than non-polluted waters (WHO 1993). Based on available PCB information and analogue information on DDT, non-polluted waters of the Great Lakes region have been estimated to contain <5 $ng \cdot L^{-1}$, moderately polluted rivers and estuaries, 50 $ng \cdot L^{-1}$, and heavily polluted rivers, 500 $ng \cdot L^{-1}$ (WHO 1993). Mean PCB concentrations in the waters of eastern Canada, Quebec and Ontario ranged from not detected to 0.04 $\mu g \cdot L^{-1}$ in the 1985 NAQUADAT database; PCBs were not detected in waters of western Canada (Strachan 1988). PCB concentrations in sediments are much higher than those in the surrounding water as a result of the hydrophobic and adsorptive nature of the compound. Use of PCBs as an antifouling agent in paints used for ship hulls has resulted in significant contamination in harbour sediments and biota (Jensen et al 1972; Maruyama et al 1983). Sediment PCB concentrations from the Niagara and St. Lawrence Rivers averaged 0.58 $\mu g \cdot g^{-1}$ and 0.18 $\mu g \cdot g^{-1}$, respectively in the 1985 NAQUADAT database (Strachan 1988). PCB levels in the Canadian aquatic environment have been extensively reviewed by Moore and Walker (1991).

Soil

Sources of PCBs to the soil environment include leakage from landfill sites, application of municipal sewage sludges to land, volatilisation from surface waters and subsequent deposition from the atmosphere, landfilling of contaminated dredged materials, incomplete degradation in waste incinerators and accidental spills during transport. Deposition from the atmosphere to the terrestrial environment occurs in the particulate form and through gas absorption to plants, soils and water (UN ECE 1994). Sewage sludge is the main repository for PCBs in the processing of municipal effluents, with levels of 76.7 $mg \cdot kg^{-1}$ (dry weight) recorded in Ontario and 2.5 $mg \cdot kg^{-1}$ in British Columbia (Strachan 1988). The important localized sources of PCB contamination, which can result in releases of PCBs to soil and goundwater, are by disposing of municipal sewage sludges and industrial

wastewater treatment sludges in landfills and by spreading them on agricultural land (Webber et al. 1981; Pupp 1985; Strachan 1988).

Soil samples from 30 agricultural fields in eight Canadian provinces and eight sludge-treated fields in Ontario were analysed for total PCB concentration (Webber and Wang 1995). PCBs were observed in all of the agricultural soils in concentrations ranging from 0.15 to 0.235 $mg \cdot kg^{-1}$ but were detected in only two of the eight sludge-treated soils; the highest level recorded was 0.514 $mg \cdot kg^{-1}$ at the Brantford landfill site (Webber and Wang 1995). Webber et al. (1981) observed that municipal sludge application substantially increased the PCB concentration of soils in Ontario. They reported total PCB concentrations ranging from 0.007 to 0.025 $mg \cdot kg^{-1}$ in ten control crop soils, whereas, elevated PCB levels in concentrations ranging from 0.103 to 0.453 $mg \cdot kg^{-1}$ were observed at five adjacent sludge-treated sites (Webber et al. 1981).

Landfill disposal of PCB-containing waste may result in the release of PCBs to groundwater. Transformers from a hydroelectric station were buried for four years and then removed at a site near McLeese Lake, British Columbia. PCB concentrations in local wells were subsequently measured at levels up to 0.20 $\mu g \cdot L^{-1}$. Elevated levels of PCBs were also recorded in nearby surface waters and fish populations (Garrett 1983 cited in Strachan 1988).

Soil samples from old urban and rural parkland sites in Ontario were analysed to determine the Ontario typical range background concentrations (OTR_{98}) of organic and inorganic compounds. For soil samples from rural parkland, 98% had total PCB concentrations of 0.015 $mg \cdot kg^{-1}$, while 98% of old urban parkland sites had total PCB concentrations of 0.032 $mg \cdot kg^{-1}$ (OMEE 1993). Although PCBs originate solely from anthropogenic sources, acceptable soil "background" concentrations have been defined for Ontario soils. The recommended acceptable background concentration that applies to agricultural and residential/parkland land uses was defined as 0.3 $mg \cdot kg^{-1}$ for total PCBs (OMEE 1994).

Terrestrial Biota

PCBs are persistent and bioaccumulate in many organisms as a result of their high lipid solubility and low biodegradability. PCBs have been measured in a wide variety of biota from many different environments. Comprehensive lists of PCBs residues in biota can be found in Risebrough et al. (1968), Peakall (1975) and Eisler (1986), while organochlorine levels in Arctic biota are reviewed in INAC (1997) and Thomas et al. (1992).

PCBs are accumulated in the lipid-rich tissues of organisms therefore care must be taken when interpreting results between species with different amounts of body fat. Animals with large amounts of body fat, such as Arctic mammals and birds, are particularly susceptible to PCB accumulation (WHO 1993). Redistribution of PCBs from fat to other tissues occurs

in animals during periods of enforced starvation (e.g., hibernation, migration, and incubation) and the feeding of offspring (WHO 1993).

PCB residues in animals tend to increase with trophic level as a result of biomagnification. For example, eggs of carnivorous birds from the Niagara peninsula contained much higher levels of PCBs than those of herbivorous or insectivorous birds (Frank et al. 1975). Similarly, PCB concentrations measured in adult polar bears and their cubs were higher than those in Arctic charfish and seals (Bowes and Jonkel 1975 cited in WHO 1993), while piscivorous and molluscivorous Arctic birds contained more PCBs than browsing and grazing birds per kg of body weight (INAC 1997).

Body burdens of PCBs may increase with age due to slow rates of biotransformation and excretion. However, in some species of mammals, the trend with age may not be observed in females as a result of the transfer of PCBs to offspring during lactation. Consequently, females may have lower body burdens of PCBs than males (Addison et al. 1986; WHO 1993).

PCB levels in terrestrial mammals may show geographic variation in some species. Mink in the southern Northwest Territories had higher PCB concentrations than those from the MacKenzie delta, but lower than those in southern Ontario (INAC 1997). A substantial west to east increase in PCBs was observed in caribou, likely as a result of the predominant west to east/north-east atmospheric circulation pattern that delivers PCBs from industrialised regions of North America (INAC 1997). Eggs of terrestrial birds collected in a rural environment in Ontario contained lower PCB levels than those sampled from urban areas (Frank et al. 1975).

PCB congener patterns may vary between species of animals. Over 95% of the total PCBs measured in polar bears *(Ursus maritimus)* were represented by congeners 99, 153, 138, 180, 170 and 194. Congeners notably absent in polar bears, though present in seals, were PCBs 74, 66, 118, 146, 105 and 187 (Norstom 1988). Similarly, Muir et al. (1988) observed that tri- and tetrachloro PCB congeners were the dominant PCBs in fish while pentachloro/hexachloro and hexachloro/heptachloro congeners predominated in ringed seal blubber and polar bear fat, respectively. Although limited data are available, PCB concentrations in polar bears have shown little change since the 1960s (Norstrom 1997).

Peregrine falcon *(Falco peregrinus)* populations have been a focus of environmental concern for several decades. An analysis of PCB residues in 205 eggs over the period 1965 - 1987 showed that PCB levels ranged from not detected to 37.5 $mg \cdot kg^{-1}$ ww. No distinct trends in PCB levels with time or spatial variation were observed (Peakall et al. 1990). An analysis of PCB residue levels in prey species of Peregrine falcons showed that PCBs can still be found in prey at levels which could be associated with deleterious effects on the reproductive success of nesting birds. The study also suggested the existence of PCB “hot spots” in the Great Lakes and southern prairie regions (Baril et al. 1990).

Hebert et al. (1996) quantified non-ortho-substituted PCBs in caribou from the Canadian Arctic and reported higher concentrations in fat tissue than in muscle or liver. Concentrations of individual congeners were relatively low (≤0.3 $ng \cdot g^{-1}$, lipid wt.). No detectable PCBs were found in fat and serum samples taken from musk oxen from the Northwest Territories (Salisbury et al. 1992). The low PCB concentration (1.2 $\mu g \cdot g^{-1}$ in fat tissues) found in the Arctic sheep *(Obis aries)* can be explained by its feeding solely on plants (Clausen et al. 1974).

Existing Environmental Quality Criteria and Guidelines for PCBs

Soil quality criteria and guidelines for PCBs have been developed by several jurisdictions, including Canada, the Netherlands and the United States (Appendix III). Jurisdictions within Canada with existing guidelines for PCBs in soils are Ontario, Quebec, Alberta and Saskatchewan. PCB guidelines for soils are usually expressed as total PCBs, although the Netherlands has also established guidelines for specific congeners. PCB guideline values from jurisdictions within Canada range from 0.5 to 25 $mg \cdot kg^{-1}$ dw, while values from countries other than Canada range from 0.02 to 100 $mg \cdot kg^{-1}$ dw.

Guidelines and criteria for PCBs in groundwater have also been developed by Canada, the United States and the Netherlands (Appendix IV). Jurisdictions within Canada with existing guidelines for PCBs in groundwater are Ontario and Quebec. PCB guidelines for groundwater are usually expressed as total PCBs. Guideline values from jurisdictions within Canada range from 0.0002 to 0.001 $mg \cdot L^{-1}$, while values from countries other than Canada range from $2x10^{-8}$ to 0.01 $mg \cdot L^{-1}$.

CHAPTER 3. ENVIRONMENTAL FATE AND BEHAVIOUR OF PCBs

Behaviour in the Atmosphere

Direct releases of PCBs to the atmosphere include vaporisation from PCB-containing products and from landfills, and incomplete incineration of PCB-containing waste (CCREM 1986; Eisler 1986; Strachan 1988; UN ECE 1994). Although emissions from anthropogenic sources have decreased, the drop in atmospheric concentrations has been slow (Buckley 1982), possibly as the result of PCB recycling by the environment (Murphy et al. 1985). There is evidence to suggest that particulate PCBs deposited to aquatic environments may dissolve and re-evaporate (Murphy et al. 1985). Fugacity modeling also indicates that some amount of the PCBs deposited to the aquatic and soil environments may evaporate and re-enter the atmosphere (Mackay et al. 1992).

Atmospheric PCBs may be in the gaseous phase or adsorbed onto particles. Airborne PCBs are often associated with suspended particulates in urban areas, but most (99%) of the atmospheric PCBs in rural areas are in the vapour phase (NRC 1979). Although atmospheric concentrations of PCBs are generally low, 18 t are calculated to be present in the U.S. atmosphere at any given time (NRC 1979). An estimated 900 t of PCBs are cycled through the U.S. atmosphere each year (Buckley 1982), representing an average input to and deposition from the atmosphere of about 60 $g \cdot km^{-2} \cdot a^{-1}$ (Murphy et al. 1985). This atmospheric reservoir represents a direct source of PCBs to the Canadian environment.

PCBs in the atmosphere may be removed by wet and dry deposition, but particle deposition appears to be the most important mechanism in near-source transport (UN ECE 1994). Congeners with more than four chlorine atoms are also subject to photodecomposition (Hutzinger et al. 1974; Strachan 1988; Moore and Walker 1991). PCBs may remain airborne over long distances (Buckley 1982). Based on limited data, the residence time of PCBs in the atmosphere has been estimated at approximately 18 days (Bidleman et al. 1976).

Atmospheric transport and deposition of PCBs are the primary mechanisms leading to the contamination of remote environments (Buckley 1982; Tanabe 1988). PCB dispersion by air depends upon the volatility of individual PCB congeners, prevailing winds and precipitation patterns, and possible photodegradation of higher chlorinated isomers. There is evidence to suggest that PCBs are continually being transported to the Arctic via the atmosphere (Hargrave et al. 1988), deposited in aquatic and terrestrial systems (Gregor and Gummer 1989) and incorporated into Arctic food webs (Muir et al. 1988). This tendency of PCBs to be transported to the poles suggests that PCBs currently sequestered in environmental matrices in southern Canada, the US and other countries could ultimately be transported to northern Canada. Thus, northern Canadian ecosystems could continue to receive substantial inputs of PCBs in spite of the legislation currently limiting their use.

Behaviour in Aquatic Systems

Deposition of atmospheric PCBs is a major source of contamination to Lakes Michigan, Superior and Huron (Murphy et al. 1985; Strachan 1988). Volatilisation from the Great Lakes has been investigated, but it is not known if these water bodies are serving as sinks or sources of PCBs to the region (Strachan 1988). Evidence suggests that particulate PCBs deposited from the atmosphere to aquatic systems may dissolve and re-evaporate (Murphy et al. 1985). PCB volatilisation rates from water bodies are complex. The volatility of PCBs depends on the vapour pressure of the individual compounds, the depth of the water body, the concentration of PCBs in the bulk water, air movement (wind) above the water, and the turbulence at the air/water interface (Strachan 1988). PCBs appear to volatilise much more readily from water than their vapour pressures would suggest. It has been theorised that hydrophobic organics accumulate at the air-water interface, resulting in more volatile behaviour of the chemicals (Strek and Weber 1982a).

PCB concentrations in the open ocean and in marine mammals living in the areas of water sampling indicate that maximum PCB contamination occurs in the mid-latitudes of the northern hemisphere (Tanabe 1988). The ocean environment is an important sink for PCBs. Of the estimated 370×10^3 t currently present in the global environment, 360×10^3 t are retained in coastal sediments and open ocean water (Tanabe 1988). Open ocean waters contain approximately 60% of the global environmental PCB load, indicating that they serve as a vast reservoir and final sink for PCBs (Tanabe 1988). The North Atlantic Ocean is the dominant sink for PCBs in the environment, while freshwater sediment is a major continental reservoir (NRC 1979). PCBs are strongly adsorbed on soils, sediments and particulates in the environment, with the highest concentrations found in aquatic sediments containing high organic or clay content (Eisler 1986).

PCBs are denser than water and tend to sink to the bottom of the water column (CCREM 1986). As a result of their physical and chemical properties, PCBs can be removed from the water column by adsorption to particulate matter that accumulates in the sediments of lakes, rivers and estuaries where they may remain available for re-suspension for at least 8 - 15 years (Swain 1983; Moore and Walker 1991). PCBs are primarily associated with suspended microparticulates with diameters of less than 0.15 mm. As well, PCB concentrations increase with organic content of these particulates. PCBs adsorbed onto sediments can be released to the surrounding water by slow desorption, especially when sediments are resuspended and redistributed during flooding and dredging events or by translocation of sediments through biological activity (Moore and Walker 1991). Desorption of individual congeners is affected by the number and placement of chlorine atoms on the biphenyl ring. For instance, coplanar PCB congeners are more efficiently adsorbed to sediments than non-coplanar congeners (Moore and Walker 1991).

In general, PCBs degrade extremely slowly in the aquatic environment (Strachan 1988). Removal from the aquatic environment is primarily the result of volatilisation to the atmosphere or burial by sediments (Strachan 1988). PCBs do not undergo most hydrolytic reactions. Modeling studies indicate that the half-life of PCBs in large lakes or oceans is

approximately 1 to 2 years as the result of photolysis (Bunce et al. 1978). However, most PCBs in the aquatic environment lie buried in sediments and are inaccessible to sunlight or volatilisation unless disturbed (Moore and Walker 1991). Desorption of PCBs from sediments is a slow process. Only 0.35% of PCBs in spiked sediments were desorbed during a 120 day test in flowing water (Halter and Johnson 1977). Microbial transformation and degradation of PCBs in sediments are limited to the lower chlorinated (less than five chlorines) congeners under aerobic conditions. Thus, half-lives of PCBs in sediments are likely to be very long, on the order of years or decades (Moore and Walker 1991).

Behaviour in the Terrestrial Environment

PCBs in soil are derived from particulate deposition, wet deposition, the use of sewage sludge as a fertilizer, spills, leaks from machinery, and leaching from landfill sites (WHO 1993). In comparison to aquatic ecosystems, the occurrence and distribution of PCB congeners in terrestrial ecosystems has received little attention (Barrie et. al. 1992). However, soil can be considered a major environmental sink for PCBs as a result of their persistence and tendency to adsorb to soil particles.

The partitioning and transformation of PCBs in the terrestrial ecosystem depends upon the chemical properties of individual congeners, the properties of the soil in which they are present and the characteristics of the plants exposed to them. Soil properties which affect the mobility of chemicals include: the amount and types of colloids present; particle size distribution; pore volume and size distribution; water content; type of ground cover; and slope. Chemical properties important to movement of chemicals in soils are water solubility, ionisability, volatility and certain structural aspects or steric effects. It is important to remember that the soil system is a dynamic system involving a continuous interplay of combinations of these factors and is never at equilibrium (Strek and Weber 1982a).

Volatilisation

Volatilisation is an important potential mechanism of PCB loss from soils (Fairbanks et al. 1987). The volatilisation of PCBs from soil is dependent upon the vapour pressures and solubilities of individual congeners, PCB soil concentration, soil adsorption reactions, the water and organic matter solubility of individual congeners, temperature and soil water content (Gan and Berthouex 1994). In general, lower chlorinated PCBs tend to be more volatile than higher chlorinated PCBs (Fairbanks et al. 1987). Vapour pressures of PCBs are reduced by their interaction with soil, mainly as the result of adsorption; thus, the factors that favour volatilisation are the opposite of those that favour adsorption (Fairbanks et al. 1987; Chou and Griffin 1986).

Experiments studying the volatilisation of Aroclor 1254 from a sand surface showed that higher chlorinated congeners were less volatile than lower chlorinated congeners. After four weeks, seven-chlorine congeners showed losses of less than 10% compared to losses of over 70% for four-chlorine congeners. Losses under wet conditions were similar in

magnitude to those under dry conditions. Losses from a soil containing higher organic matter (3.1%) were negligible (Haque et al. 1974).

The major means of loss of Aroclor 1254 from three soils in a field experiment in New Mexico was by volatilisation. Half-lives for the disappearance of PCBs from soils were estimated to range from 1.2 to 4.9 years, depending on the initial concentration of PCBs in soil and soil type. Other factors that may affect the volatilisation of PCBs from soils include temperature, wind velocity, soil moisture, depth of incorporation and photodegradation. Sewage sludge additions served to decrease PCB volatilisation (Fairbanks et al. 1987).

Adsorption

Soils, sludge solids and sediments are excellent adsorbents for PCBs as a result of their high organic content, surface area, active surface properties and functional groups (Gan and Berthouex 1994). Sorption of PCBs onto soil particles is dependent upon the degree of chlorination of individual congeners, soil type, organic matter content, soil pH, and soil moisture content. There is a very high positive correlation between the total organic content of a soil and the amount of PCBs adsorbed (Chou and Griffin 1986).

The mechanism of PCB sorption is similar to the partitioning of a non-polar solute between polar and non-polar solvents. PCB, a non-polar solute, partitions from the polar aqueous phase into soil organic matter, which acts as a non-polar solvent. PCB adsorption to mineral surfaces may also occur (Chou and Griffin 1986), particularly in low organic matter soil. Sorption of PCBs in aqueous soil solutions is a two step process: an initial fast quasi-equilibrium due to diffusion into the soil surface followed by a slower diffusion further into the organic material (Cortes et al. 1991; Paya-Perez et al. 1991). Quasi-equilibrium is reached in soil solutions after a period of several hours, but true equilibrium may not be reached for many days (Cortes et al. 1991).

In general, higher chlorinated congeners adsorb more readily onto soil particles than lower chlorinated species (Cortes et al. 1991; Gan and Berthouex 1994). Soil adsorption is also stronger for coplanar PCBs than non-coplanar congeners with the same degree of chlorination (Cortes et al. 1991; Paya-Perez et al. 1991).

Soil type is an important factor in the adsorption of PCBs onto soil particles. Experiments with soil solutions of Aroclor 1254 showed that sands and silica gels do not adsorb PCBs. Woodburn soil adsorbed the highest amount of PCB, followed by illite, montmorillonite and kaolinite clays. The high adsorptive capacity of the Woodburn soil was attributed to the presence of organic matter (Haque et al. 1974).

A large spill of Askarel Inerteen 70-30, composed of 70% Aroclor 1254 and 30% tri- and tetrachlorobenzenes, occurred in Regina, Saskatchewan in 1976. The spill was the source of considerable concern and controversy as the site of the spill was located over the aquifer serving as water supply to the city. A study of the Condie silt in the area showed that the adsorption partition coefficients for individual congeners ranged from

5 000 to 26 000 $mL \cdot g^{-1}$, indicating that PCBs will move very slowly through unfractured Condie silt (Anderson and Pankow 1986).

Leaching

The movement of PCBs in soil profiles is directly proportional to the solubility of PCBs in the leaching solvent and inversely proportional to the organic matter content of the soil. PCBs are non-polar and sparingly soluble compounds in water; therefore, the penetration of PCBs into the soil profile by water flow is limited. However, PCBs are highly mobile when leached with organic solvents; thus, PCBs and organic solvents should not be disposed of in the same landfill location (Chou and Griffin 1986).

Laboratory studies indicate that PCBs are not readily leached by percolating water. During a four month period following the addition of Aroclor 1016 to three soils, a sandy loam, a silty loam and a silty clay loam, less than 0.05% of the total mixture available was leached from the sandy loam column in the worst case. Approximately 50 to 100 L of water passed through the columns during the course of the experiment, corresponding to 50 to 100 feet of rainfall in the field (Tucker et al. 1975). Furthermore, Moza et al. (1976, cited in WHO 1993) found that PCB leaching was negligible as two years after the application of ^{14}C-labelled dichlorobiphenyl to a loamy sand soil at a rate of 1 $mg \cdot kg^{-1}$, most of the detectable PCB was in the top 10 cm of the soil and only 0.2% had reached a depth of 40 cm (Moza et al. 1976, cited in WHO 1993).

Microbial Decomposition

PCBs can be biodegraded under both aerobic and anaerobic conditions. The biochemical pathway for aerobic degradation of PCBs involves initial addition of O_2 at the 2,3-position by a dioxygenase enzyme, with subsequent metabolism to chlorobenzoic acid. Aerobic biodegradation generally metabolises the less chlorinated congeners (Gan and Berthouex 1994).

Microbial reductive dechlorination occurs under anaerobic conditions. Anaerobic micro-organisms appear to utilise chlorine as the terminal electron acceptor, adding an electron to the carbon-chlorine bond, which leads to chloride loss and subsequent hydrogen abstraction. The dechlorination process leaves the biphenyl nucleus untouched and forms less chlorinated PCBs. It is possible that anaerobic metabolism could dechlorinate some congeners to a degree that makes them available for aerobic decomposition (Gan and Berthouex 1994).

The rate of microbial decomposition of PCBs depends on the degree of chlorination and the positions of chlorine atoms (Eisler 1986). In general, the microbial degradation rate of PCBs in soils generally decreases as chlorine substitution increases (Furukawa 1982). PCBs containing two chlorines in the ortho- positions of a single ring or in each ring are particularly resistant to biodegradation, while congeners with all chlorines on a single ring are degraded faster than those containing the same number of chlorines on both rings (Furukawa 1982). The half-lives of PCBs in soil have been measured at approximately

eight days for mono- and di-chlorobiphenyls, and 12 - 30 days for tri- and tetra-chlorobiphenyls. Congeners with five or more chlorine atoms showed losses of 0 to 25% within a year (Pal et al. 1980).

CHAPTER 4. BEHAVIOUR AND EFFECTS IN BIOTA

The available information on the toxicological effects of PCBs on soil microbial processes, terrestrial plants and invertebrates, as well as mammals and birds has been reviewed and summarized in this chapter in support of the derivation of environmental soil quality guidelines. This information has been tabulated in appendices VII - XV.

PCBs are universal environmental contaminants present in most ecosystems throughout the world (WHO 1993). PCBs have been shown to be both persistent and bioaccumulative (Tanabe 1988) and evidence suggests that the present environmental burden of PCBs is being recycled by natural biogeochemical processes (Addison 1986). The amount of literature on the toxicity of PCBs is immense. Information on the effects of PCBs on terrestrial biota comes primarily from experimental studies with laboratory animals and field studies that correlate PCB exposure to changes in wildlife. The probable susceptibility of wildlife to PCBs had been reported in many cases (Tanabe 1988), and recent studies by Giesy et al. 1994, among others, make a convincing case for the correlation between environmental toxicity and exposure to PCBs. Giesy et al. 1994 present evidence for the correlation between the concentrations of total dioxin-like equivalents (TEQ) in eggs and embryolethalities and deformities in embryos and chicks of fish-eating birds. They conclude that, because the planar PCBs make significant contribution to the total TEQ, they should be emphasized in wildlife and human hazard risk assessments.

Commercial PCB Formulations

Prior to the development of improved analytical techniques, toxicity data for PCBs were available only for commercial mixtures such as Aroclor (Rice and O'Keefe 1995). The complexity of commercial PCB mixtures presents immense analytical and toxicological problems as a result of the varying levels of individual congeners and impurities present within them. Toxicological problems are further aggravated by the fact that some of the trace impurities (e.g., PCDFs) in the mixtures may be more toxicologically significant than the PCBs themselves (Ax and Hansen 1975; Addison 1986; Tanabe 1988). The composition of most PCB extracts from environmental samples does not resemble that of commercial PCB mixtures as a result of environmental redistribution of individual congeners with varying volatilisation and degradation rates (WHO 1993; Safe 1994). Thus, organisms in their natural environment are seldom exposed to unadulterated manufactured products (Rice and O'Keefe 1995) and, although realistic, data from toxicological studies of PCB mixtures do not lend themselves to unequivocal interpretation (McKinney et al. 1976).

Individual PCB Congeners

The persistence in biological tissue and toxicity to biota of individual PCB congeners is determined by the structure and positions of the chlorine atoms on the molecule as well as the number of chlorine atoms present (Lech and Peterson 1983, Metcalfe and Haffner 1995). The PCB congeners that were noted to be the most biologically active generally

have chlorine substitution in at least one of the 3,3', 4,4' positions and are not chlorinated at any of the 2 and 6 (or *ortho*) positions on the biphenyl molecule. The lack of chlorine substitution in the ortho positions allows the two phenyl rings to rotate into the same plane and so these compounds are commonly called coplanar PCBs. PCB compounds with chlorine substitution at only one ortho position may also achieve partial coplanarity (Metcalfe and Haffner 1995). The non-ortho and mono-ortho PCBs commonly found in environmental samples include congeners 28, 37, 60, 66, 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, 170, 180 and 189 (Metcalfe and Haffner 1995; Eisler and Belisle 1996). The toxicities of coplanar halogenated hydrocarbons relative to that of 2,3,7,8-TCDD, the most potent compound in this class of chemicals, are often compared through a calculated toxic equivalency factor (TEF). TEFs for coplanar PCB congeners range from 1×10^{-5} for PCB 180 to 0.1 for PCB 126 (Safe 1994; Eisler and Belisle 1996; Hoffman et al. 1996; Kennedy et al 1996). However, application of the TEFs for individual congeners must be performed with caution, as the analysis of laboratory and wildlife toxicity studies suggests that PCB mixtures show both additive (synergistic) and non-additive (antagonistic) interactions between congeners (Safe 1994).

Rodent studies have shown that some PCB congeners may be carcinogenic and may also promote the carcinogenicity of other chemicals. Immunosuppression also appears to be a congener-specific effect (WHO 1993).

Bioaccumulation

Bioaccumulation is a critical aspect of the environmental fate and behaviour of PCBs. PCBs accumulate in almost all organisms as a result of their high lipid solubility and slow rates of metabolism and elimination (WHO 1993). These characteristics permit PCBs to accumulate to relatively high levels in biota, even at low exposure rates. Thus, sustained low levels of PCBs in the abiotic environment may result in adverse chronic effects in the biota exposed to them (UN ECE 1994).

The bioaccumulation of pollutants in biological systems is complex and is dependent upon several biological, chemical and physical factors (Tanabe 1988). The accumulation of PCBs from the environment by an organism involves two opposing processes: uptake and loss. Uptake may occur through dermal, inhalation or dietary exposure, while losses may occur through defecation of undigested material, excretion of metabolic waste, and the production of young (Moore and Walker 1991). The uptake and bioaccumulation of PCB congeners in biota have been correlated with the octanol/water partition coefficient (K_{ow}) and with the adsorption characteristics of PCB congeners on surfaces. Higher chlorinated PCBs and coplanar PCBs are more likely to be bioaccumulated than lower chlorinated congeners as a result of their lower water solubilities, lower volatilities and greater resistance to biodegradation (Moore and Walker 1991).

Bioconcentration factors (BCFs) are a measure of the extent to which a chemical is concentrated in tissue compared to the level at which it was present in the media or food

source from which it was taken up. BCFs should be interpreted with caution as they represent simple ratios. The magnitude of a BCF is dependent upon the exposure concentration used to calculate it. Very low exposure concentrations tend to lead to high BCFs, as all of the PCB is likely to be absorbed, while high exposure concentrations tend to minimise BCFs (WHO 1993). In addition, it must be remembered that BCFs do not account for the bioavailability of a compound. In aquatic systems, BCFs are often very large due to the relatively low solubility and high bioavailability of PCBs in water, while PCBs in terrestrial systems have much lower bioavailabilities as a result of being adsorbed onto soil particles or organic matter.

Terrestrial Microbes

Toxicological studies of PCBs on soil micro-organisms and soil microbial processes selected according to CCME 1996a to be used in guideline derivation, and other consulted studies are presented in Appendices VII and X.

Few laboratory microbial toxicity studies were available. The effects of four PCB congeners, PCBs 5, 8, 29, and 77, and Aroclors 1242 and 1260 on microbial respiration were evaluated in Parabrownearth and Podsol soils. PCBs 5 and 8 reduced both long-term and short-term respiration in the Podsol at concentrations of 1 $mg \cdot kg^{-1}$, while PCB 29 became effective at levels of 10 $mg \cdot kg^{-1}$. No adverse effects were observed with PCB 77. All PCB congeners were less toxic in the Parabrownearth than in the Podsol, likely because of its higher clay and organic matter contents. Long-term respiration was decreased by Aroclor concentrations of 50 $mg \cdot kg^{-1}$ in the Podsol and 100 $mg \cdot kg^{-1}$ in the Parabrownearth. In general, PCB toxicity decreased with increasing chlorination for both the individual congeners and the Aroclor mixtures (Wilke and Brautigam 1992).

Soil alga *(Navicula pelliculosa)* cultured on contaminated algal assay media showed sharp decreases within 48 hours in photosynthesis rates and cell numbers at levels of 20 $\mu g \cdot L^{-1}$ of Aroclors 1221, 1242, 1016 and 1248. The order of toxicity of the mixtures varied, but Aroclor 1016 was consistently the least toxic of the four, possibly as a result of its lower content of higher chlorinated isomers (Glooschenko and Glooschenko 1975). The mycelial growth of a soil microfungi *(Aspergillus flavus)* grown in contaminated broth solutions decreased progressively as concentrations of Aroclor 1254 increased from 5 to 50 $mg \cdot L^{-1}$. Lower chlorine PCB mixtures (Aroclors 1232, 1242, 1248 and 1254) had a much greater effect on mycelial growth than a high chlorine mixture (Aroclor 1260) (Murado et al. 1976). Pyralene, a commercial mixture of trichlorobenzene and PCBs, inactivated amoebae at soil levels of 2500 $mg \cdot kg^{-1}$ (Steinberg et al. 1990).

A long term field study conducted in the vicinity of a municipal incinerator indicated that very low soil concentrations of PCBs can influence microbial processes. A plot directly impacted by emissions from the incinerator showed soil PCB levels of 0.014 $mg \cdot kg^{-1}$. Microbes taken from this contaminated plot showed significant reductions in nitrification

rates, biomass content and respiration rates when compared to a control plot with PCB concentrations of 0.0044 $mg \cdot kg^{-1}$ (considered to be a background level of contamination for the area). Levels of other contaminants such as PAHs, PCDDs, PCDFs, DDT and heavy metals were either at or below normal background concentrations in both plots and the authors concluded that the observed effects were the result of PCB contamination (Dusek 1995; Dusek and Tesarova 1996).

Terrestrial Plants

Uptake and Behaviour

The uptake of PCBs into plants is a possible first step in introducing PCBs to the terrestrial food chain and is critical to estimating potential effects to the terrestrial ecosystem. Accumulations of PCBs in plant tissues may be the result of direct root uptake from contaminated soils, localised movement of PCBs from contaminated soils in particle-bound or vapour form to plant foliage, or long-range atmospheric deposition to plant foliage (Buckley 1982; Dushenko et al. 1996).

The uptake of PCBs into plants from contaminated soil has been reported as both positively correlated with and not related to the soil concentration of PCBs (e.g. Strek and Weber 1982a; Bacci and Gaggi 1985; Webber et al. 1994). Lower chlorinated congeners are more readily taken up by plants as a result of their greater mobility in soils (Strek and Weber 1982a; WHO 1993). Roots tend to accumulate PCBs more than stems or foliage and much of this contamination consists of adsorption on the surface of the root rather than translocation within the root (Strek and Weber 1982a; Wallnöfer et al. 1975; Moza et al. 1979; Iwata and Gunther 1976). Therefore, root uptake causes little contamination of above-ground foliage; several studies have shown that the primary source of PCBs to leaves and stems is surface deposition from the atmosphere to the plant (Buckley 1982). Consequently, root crops are more susceptible to contamination by translocation or adsorption than leafy or grain crops (Gan and Berthouex 1994).

Bioconcentration factors of PCBs in plants are generally low, but vary substantially by species (Appendix V) (Strek and Weber 1982a; WHO 1993). Bioconcentration factors measured in the field for Arctic plants varied widely and ranged from 0.004 to 80 with a mean value of 3.01 (Reimer et al. 1993). It should be noted, however, that field studies cannot separate the accumulation of PCBs by root uptake and localised PCB movement from that by long-range atmospheric deposition.

There may be a minimal limit of PCB concentration in soils at which no detectable PCB is taken up by plants (Strek and Weber 1982a). For instance, 4,4'-di-, 2,2',5,5'-tetra- and 2,2',4,4',5,5'-hexachlorobiphenyls could not be detected in carrots grown in soil contaminated at a concentration of 0.05 $mg \cdot kg^{-1}$ after 120 days (Wallnöfer et al. 1975).

Toxicity

Toxicological studies of PCBs on terrestrial plants selected according to CCME 1996a to be used in guideline derivation, and other consulted studies are presented in Appendices VIII and XI.

Selected Studies

Limited data exist on the toxicity of PCBs to higher plants, particularly at soil concentrations between 100 $mg \cdot kg^{-1}$ and 1000 $mg \cdot kg^{-1}$. Documented effects of PCBs on plants include reductions in height, weight and water use (Strek and Weber 1982a). Few data were available on the effect of PCBs on seed germination rates. Beet and redroot pigweed were the most PCB-sensitive plants found in the literature, while corn and sorghum appeared tolerant of high levels of PCBs.

Weber and Mrozek (1979) evaluated the toxicity of Aroclor 1254 at concentrations of 0 $mg \cdot kg^{-1}$, 1 $mg \cdot kg^{-1}$, 10 $mg \cdot kg^{-1}$, 100 $mg \cdot kg^{-1}$ and 1000 $mg \cdot kg^{-1}$ to two crops, soybeans *(Glycine max)* and fescue *(Fescue arundinacea)*. Soybean height and fresh top weight were significantly inhibited by 15% and 22%, respectively at soil concentrations of 1000 $mg \cdot kg^{-1}$. Malformation of newly developing leaves, inhibited root growth and decreased water uptake were also observed at this concentration. The fresh top weight of fescue was significantly inhibited by 16% at soil concentrations of 1000 $mg \cdot kg^{-1}$, but no other adverse effects were observed (Weber and Mrozek 1979).

Strek et al. (1981) continued the work reported in Weber and Mrozek (1979) by evaluating the second and third plantings of soybeans in the same contaminated soils in an effort to examine the longevity of effects of soil-applied PCBs. Significant reductions in height (22% and 18%), fresh top weight (24% and 37%) and cumulative water use (47% and 32%) were observed at 1000 $mg \cdot kg^{-1}$ dose rate for the second and third soybean crops, respectively. Significant reductions (18 to 29%) in the height of the third soybean crop were observed at all application rates. The toxicity of Aroclor 1254 was also evaluated for beets *(Beta vulgaris)*, corn *(Zea mays)* and sorghum *(Sorghum bicolor)* in this study. The height, fresh top weight and cumulative water uptake were dramatically reduced in beets at the 1000 $mg \cdot kg^{-1}$ dose (100%, 100% and 96%, respectively, after 56 days). The applied doses of PCBs had no effect on the corn or sorghum plants, except for a reduction in corn height during the first five days of growth at the 100 $mg \cdot kg^{-1}$ and 1000 $mg \cdot kg^{-1}$ levels. The corn plants appeared to recover after this period.

Significant reductions in the height of redroot pigweed *(Amaranthus retroflexus)* were observed at soil Aroclor 1254 concentrations of >100 $mg \cdot kg^{-1}$. Plant height was reduced by approximately 31% at 150 $mg \cdot kg^{-1}$ and 35% at 200 $mg \cdot kg^{-1}$. The addition of organic matter and clay to the soil reduced the toxic effects by reducing the bioavailability of the soil-applied PCB (Strek and Weber 1982b). Two species of the genus *Panicum, P.*

bisulcatum and *P. virgatum*, were unaffected by soil concentrations of 100 $mg \cdot kg^{-1}$ ^{14}C-labelled Aroclor 1254 (Strek and Weber 1980).

Consulted Studies

Seed germination experiments with seventeen species of prairie grasses indicated that most species were tolerant of low concentrations of Aroclor 1260 and showed greater than 80% emergence at soil concentrations of up to 130 $mg \cdot kg^{-1}$. Exceptions included smooth brome *(Bromus inermis)*, streambank wheatgrass *(Agropyron riparium)*, crested wheatgrass *(Agropyron cristatum)* and orchard grass *(Dactylis glomerata)* which showed emergence rates of less than 10% at concentrations of 130 $mg \cdot kg^{-1}$, 110 $mg \cdot kg^{-1}$, 130 $mg \cdot kg^{-1}$ and 33 $mg \cdot kg^{-1}$, respectively. The soil sample in this experiment was obtained from a parking lot contaminated with Aroclor 1260 transformer oil and it was unclear if other toxic contaminants were present in the soil (Siciliano et al. 1997).

Saltmarsh vegetation *(Spartina alterniflora)* grown in PCB-contaminated sand (1.7 $\mu g \cdot kg^{-1}$) showed significant decreases in cumulative change in height (~30%) and cumulative number of live leaves per stem (~25%), but showed significant increases in the cumulative number of stems per plant (~300%). Alternatively, plants grown in contaminated mud (2.2 $\mu g \cdot kg^{-1}$) showed significant decreases in number of stems per plant (~75%) and altered biomass distributions (Mrozek et al. 1983). Differences in the responses of plants to PCB treatments in sand and mud are thought to be related to properties of the growth media.

A five times increase in somatic mutations was observed in ostrich ferns *(Matteuccia struthiopteris)* growing near the Housatonic River in Pittsfield, Massachusetts on sediments containing mean PCB residues of 26 $mg \cdot kg^{-1}$ (mostly as Aroclor 1254) when compared to ferns from control areas. Additionally, the Housatonic mutations were qualitatively different from the control mutations. As this was a field study, the relationship between the mutation rates and PCB contamination of the soil is correlational rather than causal and other contaminants may have been present (Klekowski 1982).

Soil samples from a contaminated salvage yard in Makinson's, Newfoundland, were found to contain PCB concentrations ranging from 28 $mg \cdot kg^{-1}$ to 120 $mg \cdot kg^{-1}$. The samples also contained elevated levels of several metals and total petroleum hydrocarbons. Toxicity tests indicated that the contaminants within the soil samples had no effect on the emergence of lettuce seedlings *(Lactuca sativa)* (Carter et al., unpublished results).

Terrestrial Invertebrates

Uptake and Behaviour

PCBs may be mobilised from the soil to the terrestrial food chain via uptake by soil invertebrates. Uptake of PCBs from soil to invertebrates may occur by direct absorption through soil contact and by soil ingestion (Belfroid et al. 1996).

The uptake of PCBs was investigated at a contaminated landfill site in the Crab Orchard National Wildlife Refuge, Illinois. Beetles *(Dyscinetus picipes)* collected at sites with soil PCB levels ranging from 25 to 7279 $mg \cdot kg^{-1}$ had mean tissue concentrations of 5.22 $mg \cdot kg^{-1}$. PCBs were not detected in the tissues of beetles collected from control areas. Bioconcentration factors could not be calculated from this study since beetles may fly some distance between daytime refuges and nighttime feeding areas and since the beetles may have had separate exposures as larvae and adults (McKee 1992).

House crickets *(Acheta domesticus)* suspended in cages over contaminated landfill soils have been shown to accumulate significant levels of PCBs compared to controls (McKee 1992; Paine et al. 1993). The crickets were not in direct contact with the soil, indicating that uptake occurred through air transfer of PCBs to the crickets or to the food and water present in the cages. PCB uptake occurred rapidly, with tissue concentrations after three days of exposure measured at 1.57 $mg \cdot kg^{-1}$ (McKee 1992). Crickets in direct contact with experimentally contaminated soil showed bioaccumulation of PCBs at all rates of exposure, resulting in whole body concentrations of 11 $mg \cdot kg^{-1}$, 48 $mg \cdot kg^{-1}$, 92 $mg \cdot kg^{-1}$, 149 $mg \cdot kg^{-1}$, and 144 $mg \cdot kg^{-1}$ for soil concentrations of 100 $mg \cdot kg^{-1}$, 250 $mg \cdot kg^{-1}$, 1000 $mg \cdot kg^{-1}$ and 2000 $mg \cdot kg^{-1}$, respectively. Bioconcentration factors ranged from 0.11 to 0.19 (Paine et al. 1993).

Specimens of *Orchesella villosa*, a litter-dwelling arthropod, collected from an oak forest in central Italy showed mean PCB concentrations of 1.67 $mg \cdot kg^{-1}$ dw. Other species of arthropods collected from the same site showed PCB levels from 0.2 to five times those of *O. villosa*. Most congeners were present in very small quantities and only PCBs 153, 170 and 180 occurred in considerable amounts. Underlying soil PCB concentrations were not reported. It was noted, however, that the relative abundance of PCB congeners were similar between species of arthropods but differed between arthropods and soil, with congeners 153, 180, 170, 201, 195, 194 and 206 being more abundant in the insects than in the soil (Lupetti et al. 1994).

Arthropods collected from experimental sites treated with a commercial sewage sludge showed low levels of bioconcentration. Soil PCB residues in the treated plots ranged from 0.019 to 0.039 $mg \cdot kg^{-1}$. Species such as aphids and leafhoppers that rarely contact soil directly showed no detectable levels of PCBs. PCB uptake was observed in larger animals, such as grasshoppers, beetles and crickets that contact the ground directly. Bioconcentration factors were reported as 1.3 for primary consumers and 0.8 for secondary consumers, although this difference was not statistically significant (Davis et al. 1981).

Earthworms form a large component of the animal biomass in soils and are a major food source for birds and terrestrial fauna. Earthworms ingest enormous quantities of soil and are in continual contact with soil. Therefore, it is not surprising that earthworms have been

shown to bioaccumulate organic pollutants such as PCBs (Larsen et al. 1992 and Belfroid et al. 1995).

The uptake, accumulation and elimination of PCBs were studied in earthworms *(Eisenia andrei)* exposed to PCB-contaminated soil with individual congener concentrations ranging from 0.0198 to 0.629 $mg \cdot kg^{-1}$. Exposure times ranged from 5 to 120 days. Uptake of PCBs from the soil was rapid and steady state tissue congener concentrations of 0.0056 to 0.197 $mg \cdot kg^{-1}$ were achieved within 10 days. Corresponding congener-specific bioconcentration factors ranged from 0.26 for PCB 180 to 0.37 for PCB 118. The relative abundance of congeners in earthworms and soil were similar. Following transfer to clean soil, the earthworms showed monophasic elimination of PCBs. Calculated half-lives for the elimination of PCB congeners ranged from 12.9 days for PCB 101 to 28.2 days for PCB 180 (Belfroid et al. 1995).

A field study conducted at two sites on the contaminated Rhine-Delta flood plain investigated organochlorine concentrations in a terrestrial food chain. Mean concentrations of total PCBs were 0.2 $mg \cdot kg^{-1}$ dw and 1 $mg \cdot kg^{-1}$ dw in soils and 3.9 $mg \cdot kg^{-1}$ and 5.1 $mg \cdot kg^{-1}$ in fat of earthworms *(Lumbricus rubellus)* at the Ochten and Gelderse Poort sites, respectively. The bioconcentration ratio of PCBs in earthworm fat to that of soil organic matter varied between 0.4 to 0.7 for PCBs 101, 118, 138 and 153. The highest ratios observed were 2.5 for PCB 193, 2.6 for PCB 111 and 3.1 for PCB 182/187. PCB residues in earthworm fat were on average 0.76 times greater than that in soil organic matter (Hendriks et al. 1995).

Earthworm uptake of PCBs was studied in nine confined disposal facilities bordering the Great Lakes. Earthworm species included in the study were *Lumbricus rubellus*, *Dendrodrilus rubidus*, *Eiseniella tetraedra*, *Aporrectodea trapezoides*, *Aporrectodea tuberculata*, *Lumbricus terrestris*, *and Allolobophora chlorotica*. The purpose of this study was to determine if the concentrations of contaminants in the earthworms were hazardous to predators; therefore, reported concentrations of contaminants in the worms refers to the whole animal including ingested soil. Soil PCB concentrations, estimated æs Aroclor 1254, ranged from below the detection limit of 0.1 $mg \cdot kg^{-1}$ dw to 1.0 $mg \cdot kg^{-1}$ dw, while earthworm concentrations ranged from below the detection limit of 0.4 $mg \cdot kg^{-1}$ dw to 1.8 $mg \cdot kg^{-1}$ dw. PCB concentrations in earthworms tended to be several times that in the surrounding soil, with an average bioconcentration factor (based on sites where PCBs were detected in earthworms) of approximately 3. Based on the results of reported laboratory toxicity studies, an earthworm concentration of 5 $mg \cdot kg^{-1}$, corresponding to a soil concentration of 1.7 $mg \cdot kg^{-1}$, was suggested to be hazardous to PCB-sensitive predators (Beyer and Stafford 1993).

The uptake of PCBs was studied in earthworms *(Lumbricus rubellus)* exposed for 48 hours to soil contaminated with 150 $mg \cdot kg^{-1}$ of the commercial mixture Askarel. Congener-specific bioconcentration factors, based on the dry weight of the worms, ranged from 0.36 for PCB 194 to 0.74 for PCB 52. In general, lower chlorinated congeners were

more readily taken up than higher chlorinated congeners, possibly as a result of their lower soil sorption. The elimination of PCBs from earthworms was studied by transferring the exposed worms to a less contaminated soil with a PCB concentration of 1.5 $mg \cdot kg^{-1}$ and monitoring them for 60 days. Half-lives for elimination ranged from 2.5 days for PCB 138 to 3.9 days for PCB 118. The similarity between elimination rates suggested that PCB metabolism is not important in earthworms (Larsen et al. 1992).

Earthworms *(Oligochaeta terrestrial)* collected near a factory lot contaminated with Aroclor 1260 were found to contain PCB levels of 1.85 to 2.07 $mg \cdot kg^{-1}$ dw. Soil concentrations of PCBs in the vicinity were in the range 0.27 to 2.4 $mg \cdot kg^{-1}$. Earthworm PCB concentrations in animals collected from an uncontaminated site averaged 0.68 $mg \cdot kg^{-1}$ (Greichus and Dohman 1980).

Kreis et al 1987 measured PCB levels in two Switzerland arable soils and in various species of earthworms. Soil total PCB concentrations varied from 1 to 13 $mg \cdot kg^{-1}$ and total PCB concentrations in depurated worms varied from 11 to 174 $mg \cdot kg^{-1}$. The authors proposed the adult earthworm *Nicodulus spp*. as a potentially interesting indicator of PCB soil contamination. Also, Diercxsens et al. (1985) reported depurated worm concentrations varying from 59 to 3588 $mg \cdot kg^{-1}$. The authors noted that earthworm tissue and gut content showed PCB concentrations with a difference in fingerprint of some single PCB compounds compared to the surrounding soil.

Toxicity

Toxicological studies of PCBs on terrestrial invertebrates, selected according to CCME 1996a to be used in guideline derivation, and other consulted studies, are presented in Appendices IX and XII.

Selected Studies

The acute toxicity of soil contaminated with Aroclor 1254 was evaluated for soil fauna including nematodes and microarthropods. Soil collected from an oak-beech forest (pH = 3.8) was contaminated at levels of 25 $mg \cdot kg^{-1}$, 250 $mg \cdot kg^{-1}$, 500 $mg \cdot kg^{-1}$ and 2500 $mg \cdot kg^{-1}$. After an exposure period of seven days, there were no observed effects on total nematode abundance or nematode trophic groups at any concentration. Total microarthropod numbers were significantly decreased (87%) at the highest level of contamination. Within the nematode trophic groups, *Prostigmata* and *Oribatida* numbers were significantly decreased at 2500 $mg \cdot kg^{-1}$ by more than 80%, while *Mesostigmata* numbers increased at a concentration of 250 $mg \cdot kg^{-1}$. The authors suggest that the LC_{50} for arthropods is likely to be similar to that reported for crickets by Paine et al. (1993) (Parmelee et al. 1997). Although the soil pH reported in this study was slightly less than 4, it was included as a selected study because so few data were available for terrestrial invertebrates.

Paine et al. (1993) evaluated the toxicity of soil contaminated with Aroclor 1254 to house crickets *(Acheta domesticus)*. Twenty-one day old nymphs were exposed to soil concentrations of 100 $mg \cdot kg^{-1}$, 250 $mg \cdot kg^{-1}$, 500 $mg \cdot kg^{-1}$, 1000 $mg \cdot kg^{-1}$ and 2000 $mg \cdot kg^{-1}$ for 14 days. The LC_{50} was calculated to be 1200 $mg \cdot kg^{-1}$, corresponding to a body concentration of approximately 150 $mg \cdot kg^{-1}$. The authors suggest a benchmark whole body concentration for mortality of 100 to 300 $mg \cdot kg^{-1}$ for terrestrial insects.

Consulted Studies

Toxicity tests conducted with soil samples from a contaminated salvage yard in Makinson's, Newfoundland containing PCB concentrations in the range 28 $mg \cdot kg^{-1}$ to 120 $mg \cdot kg^{-1}$ indicated that the contaminated soil had no effect on earthworm *(Eisenia andrei)* survival. In addition to PCBs, the samples also contained elevated levels of several metals and total petroleum hydrocarbons (Carter, Mroz, Tay and Doe, unpublished results). Similarly, contaminated soils from confined facilities containing 150 $mg \cdot kg^{-1}$ as Askarel appeared to have no toxic effects on earthworms (Larsen et al. 1992).

Although no acceptable soil toxicity studies were available, earthworms appear to be less sensitive to PCBs than terrestrial insects. Earthworms *(Eisenia veneta)* exposed to filter paper contaminated with 10 $\mu g \cdot cm^{-2}$ Aroclor 1254 showed a 65% reduction in total immune activity (Bunn et al. 1996). The immune function of manure worms *(Eisenia foetida)* were unaffected by filter paper exposure at a rate of 10 $\mu g \cdot cm^{-2}$ Aroclor 1254. Exposure to this concentration resulted in a mean whole body concentration of 2900 $mg \cdot kg^{-1}$ dw. The LC_{50} was determined to be 30.4 $\mu g \cdot cm^{-2}$, corresponding to an LD_{50} of 4500 $mg \cdot kg^{-1}$ dw (Fitzpatrick et al. 1992).

Hatch and Allen (1979) observed changes in the rasping behaviour of snails *(Cepeae nemoralis)* fed a mixture of Aroclor 1254 and 1260. PCB administration at dose rates of 0.5 to 5.0 $mg \cdot kg^{-1}$ appeared to evoke the behaviour characteristic of an inadequate supply of calcium even when it was present in abundance in the diet.

Bacteriophagus nematodes *(Acrobeloides nanus)* exposed to Aroclor 1254-contaminated agar for 10 days showed decreases in number of individuals (38%), number of eggs produced (47%) and individual weight (50%) at concentrations of 15 $\mu g \cdot dish^{-1}$ (Wasilewska et al. 1975).

Birds

Uptake and Behaviour

PCBs may be taken up by birds from contaminated food or water and stored in fatty tissues. Higher chlorinated PCBs are accumulated to a greater extent than lower

chlorinated congeners. Egg-laying females can transfer substantial amounts of PCBs to their eggs. Upon starvation, PCBs preferentially redistribute to fat and the brain, such that body burdens already present may become lethal without further intake of PCBs (WHO 1993).

Laboratory studies indicate that lower chlorinated PCBs can be dechlorinated and metabolised by some species of birds (Hamdy and Gooch 1986). Studies also show that metabolism of PCBs depends on substitution patterns (Braune and Norstrom 1989). The ability to metabolise PCBs varies between avian species (Tanabe 1988). Hepatic microsomal monooxygenase activities are lower in fish-eating birds and mammals than in other animals, resulting in a high bioaccumulative tendency in these species (Tanabe 1988).

Bioaccumulation

PCBs have been found in the tissues of top level carnivores such as the gyrfalcon and the peregrine falcon in the Arctic, as well as in game birds and waterfowl, with highest levels in piscivorous and molluscivorous species (INAC 1997). There is no data linking exposure levels (levels in diet) to whole body burdens of PCBs and the estimation of bioconcentration factors is not currently possible.

Toxicity

Toxicity studies selected for use in the derivation of soil quality guidelines according to the CCME (1996a) are presented in Appendix XIII.

A number of studies have reported the toxic effects of PCBs to avian health and reproduction. Single oral dose LC_{50}s are similar to those for mammals and range from 604 to 6000 $mg \cdot kg^{-1}$ diet (Hill et al. 1975; Stickel et al. 1984; Hill and Camardese 1986). In contrast to mammals, the acute toxicity of Aroclor mixtures to birds increases with increasing chlorination (Hill et al. 1975; Hill and Camardese 1986). Sudden heavy intake of PCBs leads to high brain residues that correlate with mortality; brain residues on the order of 300 to 400 $mg \cdot kg^{-1}$ are considered to be diagnostic of acute poisoning and death (Stickel et al. 1984; WHO 1993). Chronic PCB exposure does not result in as high brain residues; death in these cases appears to be due to edema and related symptoms (WHO 1993). Sublethal effects of PCBs on birds are varied and include: decreased growth of dosed animals and their progeny, changes in organs, reproductive effects (decreased egg production and hatchability), thyroid and pituitary changes, porphyria and changes in behaviour (Hoffman et al. 1996).

As with mammalian studies, caution must be exercised in relating Aroclor and other commercial PCB mixture feeding studies to field observations because pattern recognition techniques during analysis of total PCB concentrations suggest that Aroclor mixtures change substantially in the environment and through the food chain (Hoffman et al. 1996).

Ecologically Significant Effects

The most sensitive functional endpoint for PCB toxicity appears to be reproductive impairment as associated with egg residues (Hoffman et al. 1996). The main reproductive effects of PCBs on birds are the reduced hatchability of eggs and embryotoxicity. The effects may continue after exposure to PCBs ceases as the PCB load of the mother is transferred to the eggs. There is no evidence that PCBs cause eggshell thinning directly, although effects on the food consumption and body weight of hens may have an indirect effect on shell thickness (WHO 1993).

Most studies demonstrating the effects of PCBs on reproduction have been conducted on chickens, a species that appears to be unusually sensitive to PCBs (Hoffman et al. 1996). The most severe effects result from dosing with Aroclor mixtures in the middle of the range of chlorination (Aroclors 1232 - 1254) (WHO 1993).

The reproductive toxicity of Aroclors 1221, 1332, 1242, 1248, 1254, 1268 and 5542 were evaluated for white leghorn hens *(Gallus domesticus)* in an extensive study by Lillie et al. (1974). The PCB mixtures were fed in the diet at either or both rates of 2 $mg \cdot kg^{-1}$ and 20 $mg \cdot kg^{-1}$ for nine weeks. Aroclors 1232, 1242, 1248 and 1254 fed at 20 $mg \cdot kg^{-1}$ significantly reduced egg production, the hatchability of fertile eggs and progeny growth, with Aroclor 1248 showing the most marked effects. Aroclor 1248 also significantly increased the rate of progeny mortality. In a similar study, Lillie et al. (1975) extended this research to the effects of Aroclors 1232, 1242, 1248, 1254 and 1016 at dietary concentrations of 5 $mg \cdot kg^{-1}$, 10 $mg \cdot kg^{-1}$ and 20 $mg \cdot kg^{-1}$. Aroclors 1232, 1242 and 1248 significantly decreased the hatchability of eggs at concentrations of 10 $mg \cdot kg^{-1}$ during an eight week exposure period, and Aroclors 1242 and 1248 slightly reduced progeny growth at levels of 10 $mg \cdot kg^{-1}$. In another study, white leghorn hens exposed to dietary levels of 5 $mg \cdot kg^{-1}$ Aroclor 1254 for 28 weeks showed a 20% reduction in egg production during their second fortnight of exposure (Platonow and Reinhart 1973).

White leghorn hens fed diets containing Aroclor 1242 over a period of six weeks showed no gross toxicity symptoms, such as mortality, decreases in egg production, egg weight, shell thickness and shell weight. The hatchability of the eggs was, however, reduced by exposure to PCBs. Dietary levels of 80 $mg \cdot kg^{-1}$ completely prevented eggs from hatching by the second week of exposure, while levels of 20 $mg \cdot kg^{-1}$ reduced the hatching rate to 60% of the control rate. Eggs from hens fed at the rate of 20 $mg \cdot kg^{-1}$ had yolk residue levels of 6.2 $mg \cdot kg^{-1}$ by the sixth week of exposure (Britton and Huston 1973).

Aroclor 1248 administered in diet to white leghorn hens for 8 weeks significantly reduced egg production at levels of 20 $mg \cdot kg^{-1}$ and the hatchability of eggs at levels of 10 $mg \cdot kg^{-1}$. No effects were observed on breaking strengths of eggs or feed consumption of hens. PCBs in diet were rapidly transferred to the adipose tissue of the hens, with mean concentrations of 82.7 $mg \cdot kg^{-1}$ present after eight weeks in the 20 $mg \cdot kg^{-1}$ group. PCBs

were also readily transferred to developing eggs; mean concentrations in eggs of the 20 $mg \cdot kg^{-1}$ group were 7 $mg \cdot kg^{-1}$ after eight weeks (Scott et al. 1975).

The toxicity of commercial mixtures may be due in part to low level contamination by dibenzo-p-dioxins. The eggs of white leghorn hens administered Aroclor 1242 and 1254 at the rate of 20 $mg \cdot kg^{-1}$ in diet for 10 weeks showed significantly higher rates of embryonic mortality compared to controls. Average rates of embryonic mortality were 54.7%, 59.2% and 74.0% for Aroclor 1242, Aroclor 1254 and PCB 118, respectively. Qualitative analysis revealed that all three of the administered mixtures/compounds had dibenzo-p-dioxin contamination. Purified analogues of these compounds fed to additional hens at the same rate were not embryotoxic compared to controls (Ax and Hansen 1975).

The toxicity of specific PCB congeners was evaluated for chicks in a study by McKinney et al. (1976). PCBs 128, 136, 153, 155, and 169 were fed to chicks for three weeks at concentrations of 400 $mg \cdot kg^{-1}$. The coplanar congener PCB 169 was also fed at a rate of 100 $mg \cdot kg^{-1}$. PCB 169 was the most toxic of the congeners and caused 100% mortality at an average of 13 days when fed at 100 $mg \cdot kg^{-1}$. Adverse effects were observed for all of the congeners studied, with significant decreases in body weight exhibited by birds fed PCBs 128, 136 and 153, and significant increases in liver weight by birds fed PCBs 128, 136, 153 and 155.

Japanese quails *(Coturnix coturnix japonica)* appear less sensitive to the effects of PCBs than chickens. Japanese quails exposed to 20 $mg \cdot kg^{-1}$ Aroclor 1248 in diet for eight weeks showed no decrease in egg production or hatchability (Scott et al. 1975). High levels of PCBs in diet may result in adverse reproductive effects, however. Egg production and eggshell thickness were significantly reduced following 21 day exposures to 213.5 $mg \cdot kg^{-1}$ Aroclor 1242, 78.1 $mg \cdot kg^{-1}$ Aroclor 1254 and 62.5 $mg \cdot kg^{-1}$ Aroclor 1260 (Call and Harrell 1974). Dietary exposure to 50 $mg \cdot kg^{-1}$ Clophen A60 for three weeks resulted in increased liver weights, reduced egg breaking strengths, delayed laying and diminished laying capacity (Biessman 1982). PCBs also have the potential to affect the behaviour of wildlife. Young Japanese quails exposed to 200 $mg \cdot kg^{-1}$ Aroclor 1254 showed a decreased avoidance response to moving targets after eight days of exposure. The importance of avoidance behaviour in response to the threat of predators in the wild is significant (Kreitzer and Heinz 1974).

Mallard ducks *(Anas platyrhynchos)* were also found to be relatively tolerant of PCBs. Adult pairs fed 150 $mg \cdot kg^{-1}$ Aroclor 1242 for 12 weeks before and during laying showed no difference in time taken to lay, fertility, embryo mortality, hatching success, survival rate or growth rate of chicks compared to controls. The only apparent effect of the PCBs was a 9% decrease in eggshell thickness of the dosed adults (Haseltine and Prouty 1980). Laying pairs fed diets containing 25 $mg \cdot kg^{-1}$ Aroclor 1254 for a month prior to laying had no effect on the number of hens laying, time to first laying, clutch size, fertility, nest attentiveness, number of hatchlings or hatchling survival (Custer and Heinz 1980).

PCBs have a pronounced effect on mourning dove *(Zenaida macroura carolinensis)* courtship behaviour. Doves fed 10 $mg \cdot kg^{-1}$ Aroclor 1254 for 42 days prior to courting spent twice as much time in the courtship phase as controls and only half of the dosed pairs progressed into the nesting phase. Egg laying was also delayed by exposure to PCBs (Tori and Peterle 1983). The incubation behaviour of ring doves *(Streptopelia risoria)* may also be affected by dietary exposure to PCBs. Diets containing 10 $mg \cdot kg^{-1}$ Aroclor 1254 were fed to doves for an unspecified period. Nest attentiveness of the dosed group declined, with the mean temperature of the eggs dropping from 37.3°C to 36.1°C. The daily profile of egg temperatures differed between experimental and control birds, with the eggs of the dosed birds showing a much greater variability in temperature. Embryonic mortality increased when the eggs were incubated by the dosed parents and decreased with artificial incubation. The authors suggest that the observed higher rate of embryonic mortality of the dosed birds was the result of erratic incubating behaviour (Peakall and Peakall 1973). It has been demonstrated that PCBs can significantly affect the brain chemistry of ring doves. Birds fed 10 $mg \cdot kg^{-1}$ Aroclor 1254 showed depressed levels of the neurotransmitters dopamine and norepinephrine. Depletion of these chemicals can result in abnormal behaviour and may partially explain the erratic courtship and incubation behaviour described above (Heinz et al. 1980). Exposure to PCBs can also result in serious reproductive effects in ring doves. Pairs of birds fed 10 $mg \cdot kg^{-1}$ Aroclor 1254 for nine months showed dramatic reductions in number of eggs hatched as a result of embryonic mortality. The dosed pairs also had reduced egg production and few of their hatchling survived to time of fledging (Peakall et al. 1972).

Few data were available on the effects of PCBs to secondary avian consumers. American kestrels *(Falco sparverius)* fed diets containing 3 $mg \cdot kg^{-1}$ Aroclor 1248 for six months showed small but significant reductions in eggshell thickness and shell weight (Lowe and Stendell 1991). Dietary exposure to Aroclor 1254 at concentrations of 33 $mg \cdot kg^{-1}$ over two months resulted in significant decreases in the number of sperm per ejaculate for American kestrels. No effect was observed on sperm motility. A highly significant interaction effect between Aroclor 1254 and Mirex was noted in this study and the semen collected from birds exposed to both chemicals was of poor quality by the end of the study (Bird et al. 1983). Aroclor 1248 fed at a rate of 3 $mg \cdot kg^{-1}$ in diet to screech owls *(Otus asio)* for eight weeks had no effect on eggshell thickness, number of eggs laid, number of hatchlings or number of fledglings (McLane and Hughes 1980).

A field study was conducted on the flood plain of the Housatonic River, Massachusetts, to assess the effects of PCB-contaminated soils and sediments on the reproductive success of songbirds nesting in the area. Clutch size, number of eggs hatched and hatching success for eight species of birds within the target population (those living within the flood plain) were compared to those of reference populations nesting outside the flood plain and to ranges reported in the literature. The eight species evaluated were the American redstart *(Setophaga ruticilla)*, barn swallow *(Hirundo rustica)*, Eastern phoebe *(Sayornis phoebe)*, rose-breasted grosbeak *(Pheucticus ludovicanus)*, American robin *(Turdus migratorius)*, red-winged blackbird *(Agelaius phoenicus)*, wood thrush *(Hylocichla*

mustelina) and yellow warbler *(Dendroica petechia)*. PCB concentrations in the upper 15 cm of the floodplain soils ranged from non-detectable to 75 $mg \cdot kg^{-1}$ with a mean of approximately 12 $mg \cdot kg^{-1}$, while sediment levels ranged from non-detectable to 200 $mg \cdot kg^{-1}$ with a mean of approximately 33 $mg \cdot kg^{-1}$. Clutch sizes, number of young hatched and hatching success were not significantly different between the target and reference populations and clutch sizes were well within normal ranges reported in the literature (Henning et al. 1997).

Mammals

Uptake and Behaviour

PCB mixtures and individual congeners may be taken up by mammals via oral, inhalation and dermal exposure. In general, PCBs are rapidly absorbed, particularly by the gastrointestinal tract after oral exposure, and are transported by the blood to the liver and adipose tissue of various organs (WHO 1993; Matthews 1983). There is also evidence of placental transport, fetal accumulation and distribution to milk (Ringer 1983; WHO 1993).

The excretion of PCBs depends on their metabolism to more polar compounds. PCBs are metabolised through hydroxylation and conjugation with thiols and other water-soluble derivatives (WHO 1993). The rates of metabolism depend upon both the number and position of chlorine atoms on the molecule, with the higher chlorinated congeners more slowly metabolised than lower chlorinated congeners (WHO 1993). Congeners that cannot be metabolised tend to concentrate in lipid-rich tissues (Matthews 1983). The persistence of individual congeners in tissues varies and is not always correlated with toxicity as differences in toxicity may be associated with specific metabolites and/or intermediates (WHO 1993).

Bioaccumulation Factors - Lab Studies

PCBs are almost universally present in the environment and are readily bioaccumulated by mammals. Biomagnification in food chains has also been demonstrated (WHO 1993). In general, higher chlorinated congeners tend to bioaccumulate more than lower chlorinated congeners (WHO 1993).

The whole body residue of PCBs increased significantly with each generation of exposure in oldfield mice *(Peromyscus polionotus)* maintained on a diet containing 5 $mg \cdot kg^{-1}$ of Aroclor 1254 for three generations. First, second and third generation mice had mean whole body concentrations of 3.28 $mg \cdot kg^{-1}$, 7.51 $mg \cdot kg^{-1}$ and 14.69 $mg \cdot kg^{-1}$, corresponding to feed-to-mice bioaccumulation factors of 0.66, 1.5 and 2.9, respectively (McCoy et al. 1995).

The whole body residues of big brown bats *(Eptesicus fuscus)* fed mealworms containing 9.4 $mg \cdot kg^{-1}$ Aroclor 1254 for 37 days ranged from 29 to 121 $mg \cdot kg^{-1}$. The corresponding mealworm-to-bat bioaccumulation factor was 6.6 (Clark and Prouty 1977). Pregnant big brown bats fed mealworms containing 6.36 $mg \cdot kg^{-1}$ Aroclor 1260 for 18 to 28 days showed mean residue levels of 20.3 $mg \cdot kg^{-1}$. Levels in their offspring averaged 4.38 $mg \cdot kg^{-1}$. These concentrations were ten times higher than those in control animals and represent a mealworm-to-bat bioaccumulation factor of 3.2 (Clark 1978).

Bioaccumulation Factors - Field studies

White-footed mice *(Peromyscus leucopus)* at a contaminated landfill site in the Crab Orchard National Wildlife Refuge, Illinois were found to have substantial body burdens of PCBs. The mean soil PCB concentration in the area was 1244 $mg \cdot kg^{-1}$ ww. The mean concentration of PCBs in the mice studied was 11.16 $mg \cdot kg^{-1}$ ww; one mouse had PCB residues of 22.1 $mg \cdot kg^{-1}$ ww (McKee 1992).

Elevated PCB body burdens were observed in white-footed mice *(Peromyscus leucopus)* living in the area surrounding Tyler Pond near Ann Arbor, Michigan. The surface water, ground water and soil surrounding the pond are heavily contaminated with PCBs and metals, with sediment PCB levels ranging from 0.1 to 290 $mg \cdot kg^{-1}$ and with a mean of 118 $mg \cdot kg^{-1}$. Adult males trapped in the area had whole body PCB levels ranging from 0.42 to 4.17 $mg \cdot kg^{-1}$ with a mean level of 2.5 $mg \cdot kg^{-1}$ (Batty et al. 1990).

A field study conducted in the Canadian Arctic examined the biomagnification of PCBs in the soil-plant-lemming food chain. Average soil PCB concentrations were much higher than plant concentrations, but plant-lemming food chain biomagnification was indicated by a 6.5-fold increase in PCB concentrations from plants to lemmings *(Dicrostonyx groenlandicus)*. The average whole-body PCB concentration in the lemmings was 0.061 $mg \cdot kg^{-1}$ (INAC 1997). An examination of the lichen-caribou-wolf food chain in the Canadian Arctic also revealed the biomagnification of PCBs. Levels of PCBs in wolves were found to be as much as an order of magnitude higher than those in caribou and lichen (INAC 1997).

The bioaccumulation of PCBs in shrews *(Crocidura fussula* and *Sorex araneus)* was observed in a field study on the Rhine-Delta flood plain. Total PCB concentrations in liver fat averaged 9.3 $mg \cdot kg^{-1}$ at the Ochten site and 130 $mg \cdot kg^{-1}$ at the Gelderse Poort site. The average bioaccumulation factors from earthworm fat to shrew liver fat ranged from three to six for various individual PCB congeners (Hendriks et al. 1995).

Bioaccumulation of PCBs was observed in voles (*Microtus* sp.), field mice (*Permuscus* sp.) and ground squirrels (*Spermophilus* sp.) collected near a factory lot contaminated with Aroclor 1260. PCB concentrations ranged from 4.85 to 17.2 $mg \cdot kg^{-1}$ in liver tissues and 3.42 to 6.87 $mg \cdot kg^{-1}$ in muscle tissues. On average, these concentrations were 2.5 times

higher than those measured at an uncontaminated site. Whole body concentrations were not recorded (Greichus and Dohman 1980).

The uptake of PCBs was studied in cows grazing at several locations on the Rhine-Delta flood plain. Soil concentrations of individual PCB congeners ranged from not detected to 0.01 $mg \cdot kg^{-1}$ dw, while milk fat concentrations ranged from not detected to 0.0053 $mg \cdot kg^{-1}$ fat weight. The bioaccumulation from soil organic matter to milk fat varied between 0.01 to 0.1 for persistent congeners (Hendriks et al. 1996).

Toxicity

Toxicity studies selected for use in the derivation of soil quality guidelines according to the CCME (1996a) are presented in Appendix XIV.

Exposure to PCBs elicits a variety of effects in mammals. The major sites of PCB pathology in mammals are the skin and liver, but the GI tract, immune system and nervous system may also be affected (WHO 1993). Common manifestations of PCBs exposure include: hepatoxicity (hepatomegaly, necrosis); immunotoxicity (atrophy of lymphoid tissues, suppressed antibody response); neurotoxicity (impaired behaviour and development, catecholamine alterations); reproductive effects (increased abortion, low birth weight, embryolethality, teratogenicity); gastrointestinal ulceration and necrosis; bronchitis; dermal effects; weak mutagenicity at high doses; and preneoplastic changes at low doses (Eisler and Belisle 1996). PCBs have been implicated as possible carcinogens and cancer promoters (Batty et al. 1990; WHO 1993). The induction of hepatic microsomal enzymes is one of the earliest and most sensitive responses to PCBs (Eisler and Belisle 1996). PCB-induced toxicity patterns vary greatly between species as a result of differing abilities to metabolise PCBs and differing primary sites of action. The toxicity of PCBs to mammals may also depend upon: the age, growth rate, mass and lipid content of the animal; dose rate, route and duration of exposure; the presence of specific congeners within the administered dose; and interactions with other compounds (Eisler and Belisle 1996).

A substantial part of the toxicity of commercial PCB mixtures results from the presence of coplanar congeners. These compounds have been demonstrated to cause toxic effects similar to those of 2,3,7,8-TCDD, including oedema, weight loss, hepatic and thymus changes, embryotoxicity, teratogenicity and immunotoxicity, in both mammals and birds (Eisler and Belisle 1996; Hoffman et al. 1996).

Acute oral LD_{50} values for PCBs in rats range from 1010 $mg \cdot kg^{-1}$ bw for Aroclor 1254 to 4250 $mg \cdot kg^{-1}$ bw for Aroclor 1242 (INAC 1997), while LD_{50} values for mink range from 750 to 4000 $mg \cdot kg^{-1}$ bw (Aulerich and Ringer 1977). Single-dose mortality data has not been reported for other mammalian species. In contrast to birds, the acute lethality of Aroclor mixtures to mammals is inversely related to chlorine content (WHO 1993). Variation in LD_{50} values may be related to the species, age and/or sex of the animal

tested, as well as the purity of the PCB formulation (ATSDR 1995). The principal signs of acute toxicity in rats include diarrhoea, respiratory depression and dehydration (ATSDR 1995).

Ecologically Significant Effects

There are few data that can be used directly to assess quantitative effects on wild terrestrial mammals at the population level (Linzey 1987). The toxicity of PCBs have been assessed for white-footed mice (Merson and Kirkpatrick 1976; Sanders and Kirkpatrick 1977; Linzey 1987, 1988), oldfield mice (McCoy et al. 1995), big brown bats (Clark and Prouty 1977; Clark 1978), mink (Aulerich and Ringer 1977; Bleavins et al. 1980; Aulerich et al. 1985, 1987; Wren et al. 1987), ferrets (Aulerich and Ringer 1977) and cottontail rabbits (Zepp et al. 1974; Zepp and Kirkpatrick 1976). Reproductive dysfunction at dietary PCB levels equal to or less than 10 $mg \cdot kg^{-1}$ have been reported for mink (Aulerich and Ringer 1977; Bleavins et al. 1980; Aulerich et al. 1985, 1987; Wren et al. 1987), white-footed mice (Linzey 1987, 1988) and oldfield mice (McCoy et al. 1995). Most other PCB toxicity data exists for standard laboratory animals.

Several studies have reported the effects of PCBs on the reproductive success of mammals. In general, reproductive toxicity is observed at doses producing systemic toxicity in the mother (WHO 1993). Neonates feeding on contaminated mother's milk are particularly sensitive to PCBs and show reduced growth as well as other toxic symptoms (WHO 1993).

The effects of chronic dietary exposure of PCBs were investigated in three generations of oldfield mice *(Peromyscus polionotus)* maintained on a diet containing 5 $mg \cdot kg^{-1}$ Aroclor 1254. Birth and weaning weights were significantly lower in first and second generation offspring of PCB-exposed animals than those of controls. Second generation offspring also exhibited a lower birth rate, longer intervals prior to the birth of the first litter and decreased survival to weaning relative to controls. The results of this study clearly show that chronic exposure to PCBs through diet has the potential to reduce fertility, growth and survival in *Peromyscus* and that these effects can be amplified through multigenerational exposure (McCoy et al. 1995).

White-footed mice *(Peromyscus leucopus)* were fed a diet containing 10 $mg \cdot kg^{-1}$ Aroclor 1254 for 9 to 18 months. First generation mice exposed to PCBs at 16 weeks of age successfully weaned significantly fewer offspring than control mice, while mice first exposed at 12 weeks of age exhibited longer intervals between births, smaller litter sizes at birth and smaller litter sizes at weaning than controls (Linzey 1987). Second generation mice exhibited poor reproductive success compared to second generation controls and the parent generation (Linzey 1988). These results suggest that exposure to PCB-contaminated foods can contribute to declines in natural populations of white-footed mice by reducing the number of young mice entering the breeding population (Linzey 1987).

Big brown bats *(Eptesicus fuscus)* were fed a diet of mealworms reared in wheat bran containing 10 $mg \cdot kg^{-1}$ Aroclor 1254. The average concentration of PCBs in the mealworms was 9.4 $mg \cdot kg^{-1}$. After an exposure period of 37 days, the weights of the exposed bats had increased significantly less than those of the control bats. During subsequent starvation, the PCB-exposed bats lost weight significantly more slowly than controls, suggesting that PCBs slow the metabolism of bats (Clark and Prouty 1977). In a similar study, pregnant big brown bats were fed mealworms raised on wheat bran containing 10 $mg \cdot kg^{-1}$ Aroclor 1260; the average concentration of PCBs in the mealworms was 6.36 $mg \cdot kg^{-1}$. The bats were maintained on this diet from the time of capture until parturition, a period of 18 to 28 days. Aroclor 1260 was found to have no effect on parent weight, litter weight or offspring mortality (Clark 1978).

The effects of dietary PCB exposure on cottontail rabbits *(Sylvilagus floridanus)* were examined in a study where the animals were maintained on a diet containing 10 $mg \cdot kg^{-1}$ Aroclor 1254 for 12 weeks. PCBs administered at this level had no effect on body weight or reproduction (Zepp and Kirkpatrick 1976).

Ferrets *(Mustela putorius furo)* were exposed to diets contaminated with 20 $mg \cdot kg^{-1}$ of either Aroclor 1242 or 1016 over a period of eight months. This level of exposure had no effect on mortality on the ferrets, but resulted in the complete reproductive failure in animals fed Aroclor 1242. The reproductive success of animals exposed to Aroclor 1016 was not significantly different from controls (Bleavins et al. 1980).

Though not strictly terrestrial, toxicity data for mink *(Mustela vison)* are included in Appendix XIV as they are among the mammals most sensitive to PCB toxicity. Mink maintained on a diet containing 20 $mg \cdot kg^{-1}$ Aroclor 1242 for eight months experienced 100% mortality (Bleavins et al. 1980). In general, the acute toxicity of Aroclor mixtures for mink decreases with increasing chlorination (WHO 1993). The reproductive failure of mink exposed to PCBs is well-documented (Aulerich and Ringer 1977; Bleavins et al. 1980; Aulerich et al. 1985, 1987). PCBs may be transferred to offspring through the placenta and lactation (Ringer 1983). Aroclor 1254 had the highest reproductive toxicity of the commercial mixtures, with significant adverse effects occurring at dietary levels of 2 $mg \cdot kg^{-1}$ when fed over a period of nine months (Aulerich and Ringer 1977). The most toxic PCB congener is PCB 169, which caused 50% mortality when fed at a level of 0.05 $mg \cdot kg^{-1}$ over 135 days (Aulerich et al. 1987).

A series of toxicity studies, summarized in WHO (1993), were consulted in order to evaluate the potential risk of carcinogenicity and cancer promotion by commercial PCBs mixtures to small mammals (see Appendix XV). The mice and rats were fed diets containing 1 to 1000 mg PCBs$\cdot$ kg^{-1} diet from various commercial mixtures (e.g. Aroclor, Kanechlor), for periods ranging from 2 to 21 months. Incidences of hyperplasic nodules, carcinomas and adenofibrosis in rats and mice were thought to result from direct PCB exposure. However, these carcinogenic effects were generally only associated with high PCB mixture concentrations (>100 mg$\cdot$ kg^{-1}). Neoplasms were both promoted and

inhibited in rodent livers by the presence of PCBs, when administered with known carcinogens. The factors which are believed to influence cancer promoting activity in mammals, include: induction of MFO activity, high level thresholds to PCBs, synergistic interactions between non-promoter and promoter PCB congeners, the presence and abundance of ortho-substituted congeners and the degree of PCB chlorination (WHO 1993). The sequence of events in administering PCBs and carcinogenic substances in animal studies are also thought to affect the promotional activity of PCBs. The presence of dibenzofuran impurities and variations in congener composition, among commercial mixtures, further complicate the interpretation of these toxicity studies (WHO 1993). For information concerning the potential carcinogenic hazard of environmental PCBs to humans, the reader is directed to Hayes (1987).

CHAPTER 5. DERIVATION OF ENVIRONMENTAL SOIL QUALITY GUIDELINES

The derivation of environmental soil quality guidelines for PCBs is outlined in the following sections for four land uses: agricultural, residential/parkland, commercial, and industrial. These guidelines have been developed according to the protocol described by the CCME (CCME 1996, 1997, and reprinted in 1999), but with some modifications.

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

Because PCBs are persistent, subject to long-range transport, and have a strong tendency to bioaccumulate and bioconcentrate in the food chain, the concept of land uses as envisioned in CCME (1996a) does not provide adequate protection for all ecological receptors. Therefore, the derivation of soil quality guidelines for residential/parkland and agricultural land uses will include the use of models that extend from and expand beyond the CCME 1996 Protocol to protect secondary and tertiary consumers from ingestion of contaminated soil and food. These models are not described *per se* in the CCME 1996 Protocol but are based on equations presented in CCME 1996a for the derivation of guidelines to protect primary consumers from adverse effects due to ingestion of contaminated soil and food. For commercial and industrial land uses, the soil quality guidelines will be based on direct soil contact. However, in site-specific situations where the size and/or location of these land uses may impact higher level consumers, the soil and food ingestion guidelines are recommended (refer to CCME 1996b for guidance on establishing site-specific objectives).

All data selected for use in guidelines derivation have been screened for ecological relevance and are presented in Appendices VII to IX, XIII, and XIV. Additional consulted toxicological studies are presented in Appendices X to XV. Studies may be excluded from use because of one or more of the following reasons:

- 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.

The relevance of applying these individual criteria to toxicity studies depends on whether in-soil tests or mammalian/avian tests are considered. Effective concentration (EC) data used in the following derivations were considered to be statistically significant according to the study from which the data were taken.

Environmental Soil Quality Guideline for Soil Contact (SQGsc)

The derivation of the Soil Quality Guideline for Soil Contact (SQG_{SC}) is based upon toxicological data for vascular plants and soil invertebrates. The toxicological data for plants and invertebrates selected according to CCME (1996a) are presented in Appendices VIII and IX.

Toxicological data were available only for the commercial mixture Aroclor 1254. However, Aroclor 1254 mixtures contain relatively high abundances of the PCB congeners 52, 77, 90, 95, 101, 110, 118, 138, 149, 153, 180 (Schultz et al. 1989). These congeners are often found dominating environmental samples: relatively high abundances of these congeners were detected in soil samples (Bright et al. 1995; Hendriks et al. 1995), vascular plants (Dushenko et al. 1996) and earthworms (Hendriks et al. 1996), from areas that were not directly impacted by point sources of Aroclor 1254 contamination. Therefore, the guidelines derived from the selected toxicological data (Appendices VIII and IX) should be regarded as guidelines for <u>total</u> PCBs present in soil.

Although several acceptable plant and invertebrate studies were available, the toxicity data set was skewed to effects concentrations on the order of 1000 mg PCBs·kg^{-1} soil and very few data points were available for concentrations less than this value. Thus, there were insufficient data to derive a soil contact guideline according to CCME (1996a). The calculation of a provisional soil contact guideline is presented below.

An examination of the data presented in Appendices VIII and IX revealed the presence of only three data points at concentrations less than 1000 mg PCBs·kg^{-1} soil. The lowest data point, 1 mg PCBs·kg^{-1} soil which was linked with a 20% reduction in soybean height, may be regarded as an outlier since other experiments with soybeans associated this level of response in plant height to concentrations of 1000 mg PCBs·kg^{-1} soil (Weber and Mrozek 1979; Strek et al. 1981). The next lowest data point was 100 mg PCBs·kg^{-1} soil, at which level corn showed decreased growth (30%) within the first five days after seedling emergence (Strek et al. 1981). This data point was used to calculate the provisional soil contact guideline for plants and soil invertebrates.

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

TEC = lowest EC/UF

where:

TEC = threshold effects concentration (mg total PCBs·kg^{-1} soil);
lowest EC = lowest effects concentration (mg total PCBs·kg^{-1} soil);
UF = uncertainty factor.

CCME (1996a) gives general guidance on criteria justifying uncertainty factors and recommends that the uncertainty factor be between 1 and 5. An uncertainty factor of three was applied to account for the very limited availability of acceptable invertebrate toxicity studies and soil contact effects data for soil PCB concentrations on the order of 100 mg total PCBs·kg^{-1} soil or less, and for the lack of toxicity data for other commercial PCB mixtures and individual congeners.

Thus,

TEC = 100 / 3 = 33.3 ≈ 33 mg total PCBs·kg^{-1} soil

Nutrient and Energy Cycling Check

There were insufficient acceptable microbial data to perform a nutrient and energy cycling check according to CCME (1996a). However, changes to microbial nitrification and respiration rates were observed in a field study at soil PCB concentrations as low as 14 ng total PCBs·kg^{-1} soil (Dusek 1995; Dusek and Tesarova 1996) (Appendix VII).

Since data were insufficient, these observations were not included in the guideline derivation. Thus, the SQG_{SC} for total PCBs was set to 33 mg total PCBs·kg^{-1} soil.

Environmental Soil Quality Guidelines for Soil and Food Ingestion

Bioaccumulation is the process by which chemical compounds are taken up by terrestrial organisms either directly from contaminated soil, water or air, or indirectly through the consumption of contaminated food. Biomagnification is the process by which tissue concentrations of accumulated chemical compounds are passed up through two or more trophic levels so that tissue residue concentrations increase systematically as trophic level increases (CCME 1996a). Care must be exercised when dealing with known or potential bioaccumulating/biomagnifying substances to ensure that concentrations of the contaminant in soil are protective of organisms at all trophic levels. The bioaccumulation and biomagnification potentials of PCBs are well documented in the scientific literature.

The specific pathways through which hydrophobic organic contaminants such as PCBs accumulate and transfer up terrestrial food chains are important factors to consider when determining the soil concentrations of these substances that are protective of organisms at all trophic levels (Jongbloed et al. 1996). Therefore, three exposure scenarios, with different trophic level receptors, will be modeled in the following sections to determine soil

PCB concentrations, which should not be exceeded in order to protect organisms from adverse effects due to ingestion of contaminated soil and food.

Soil Quality Guideline for Food and Soil Ingestion for the Protection of Primary Consumers (SQG_{1C})

Primary consumers (herbivores) may be exposed to large quantities of contaminated forage and soil. To evaluate the potential levels of soil-associated PCBs likely to cause an adverse effect on primary consumers, the model described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996a) was used. This model estimates the soil concentration (SQG_{1C}) that should not be exceeded in order to prevent adverse toxicological effects upon primary consumers. The procedure accounts for the bioconcentration of a contaminant from soil to plant for the protection of herbivorous animals and may also be used for the bioconcentration from soil to invertebrate for the protection of insectivorous animals.

Several assumptions are made in modeling the soil-plant-herbivore pathway:

- 100% of the contaminant burden in the herbivore is accumulated through the ingestion of contaminated soil and food.
- The herbivore remains on the contaminated site 100% of the time.
- 100% of the food ingested by the herbivore is consumed from a PCB contaminated site.

According to the data reported in the literature, the most sensitive herbivorous bird or mammalian species is the white leghorn chicken (*Gallus domesticus*) which exhibited biologically significant adverse reproductive effects (20% decline in egg production) when fed 5 mg Aroclor 1254·kg^{-1} in diet (Platonow and Rheinhart 1973). Assuming that the white leghorn has a body weight of 1.95 kg and a food consumption rate of 0.132 kg·d^{-1} (US EPA 1988), an effects dose of 0.34 mg PCB·kg^{-1} bw·d^{-1} can be calculated from the reported dietary concentration.

This effects dose (ED_{1C}, where the subscript 1C stands for primary consumer) is used to calculate the daily threshold effects dose ($DTED_{1C}$) according to the equation:

$$DTED_{1C} = \text{lowest } ED_{1C}/UF$$

where:

$DTED_{1C}$	=	daily threshold effects dose for the primary consumer (mg PCB·kg^{-1} bw_{1C}·d^{-1});
lowest ED_{1C}	=	lowest effects dose (mg PCB·kg^{-1} bw_{1C}·d^{-1});
UF	=	uncertainty factor (if needed); no uncertainty factor was applied since mammalian toxicity data in the literature was adequate (CCME 1996a).

Therefore, $DTED_{1C}$ = 0.34 mg total PCBs·kg^{-1} bw_{1C}·d^{-1}

An animal may be exposed to an environmental 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 $DTED_{1C}$. However, since PCB compounds are highly hydrophobic, and therefore have a strong tendency to partition to lipids or organic carbon, it is assumed that 100% (1.00) of exposure can be attributed to the ingestion of food and soil for a terrestrial food chain. It follows then, that exposure from soil and food ingestion should not exceed the $DTED_{1C}$.

$$\text{Exposure from direct soil ingestion + exposure from food ingestion} = 1.00 \cdot DTED_{1C}$$

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. The animal's soil ingestion rate is calculated as a proportion of the DMIR according to the equation:

$$SIR_{1C} = DMIR_{1C} \cdot PSI_{1C}$$

where:

SIR_{1C}	=	soil ingestion rate of the primary consumer (kg dw soil·kg^{-1} bw_{1C}·d^{-1});
$DMIR_{1C}$	=	dry matter intake rate of the primary consumer (kg dry matter·kg^{-1} bw_{1C}·d^{-1});
PSI_{1C}	=	soil ingestion proportions of the primary consumer reported with the DMIR.

Given the body weight and the food consumption rate mentioned above, the dry matter intake rate ($DMIR_{1C}$) for the white leghorn has been calculated to be 0.068 kg dry matter·kg^{-1} bw_{1C}·d^{-1}, with a maximal PSI value of 0.077 (McMurter 1993).

SIR_{1C} = (0.068 · 0.077) = 0.005 kg dw soil·kg^{-1} bw_{1C}·d^{-1}

The SIR_{1C} can then be combined with the concentration of the contaminant in the soil (SQG_{1C}) to calculate the exposure from soil ingestion.

$$\text{Exposure from soil ingestion} = SIR_{1C} \cdot SQG_{1C}$$

where:

SIR_{1C} = soil ingestion rate of the primary consumer (kg dw soil·kg^{-1} bw_{1C}·d^{-1});
SQG_{1C} = concentration of the contaminant in soil protective of primary consumers (mg PCB·kg^{-1} soil).

Exposure from food ingestion

Similarly to the SIR_{1C}, the food ingestion rate (FIR) of livestock and wildlife is expressed as a portion of the DMIR minus the soil ingestion rate. The FIR is calculated as:

$$FIR_{1C} = DMIR_{1C} - SIR_{1C}$$

where:

FIR_{1C} = food ingestion rate (kg dw food·kg^{-1} bw_{1C}·d^{-1});
$DMIR_{1C}$ = dry matter ingestion rate (kg dry matter·kg^{-1} bw_{1C}·d^{-1});
SIR_{1C} = soil ingestion rate (kg dw soil·kg^{-1} bw_{1C}·d^{-1}).

Thus, FIR_{1C} = 0.068 - 0.005 = 0.063 kg dw food·kg^{-1} bw_{1C}·d^{-1}.

The FIR_{1C} can be combined with the bioconcentration factor from soil to plant (BCF_1) and the concentration of the contaminant in the soil (SQG_{1C}) to represent the exposure from food ingestion.

$$\text{Exposure from food ingestion} = FIR_{1C} \cdot BCF_1 \cdot SQG_{1C}$$

where:

FIR_{1C} = food ingestion rate of the primary consumer (kg dw food·kg^{-1} bw_{1C}·d^{-1});
BCF_1 = bioconcentration factor from soil to plant (dw:dw);
SQG_{1C} = concentration of the contaminant in soil protective of primary consumers (mg PCB·kg^{-1} dry soil).

Exposure from direct soil ingestion and food ingestion

The equations from soil ingestion and from food ingestion can be combined and rearranged to solve for the SQG_{1C}.

$$(SIR_{1C} \cdot SQG_{1C}) + (FIR_{1C} \cdot BCF_1 \cdot SQG_{1C}) = 1.00 \cdot DTED_{1C}$$

Rearranging and solving for SQG_{1C}:

$$SQG_{1C} = \frac{1.00 \cdot DTED_{1C}}{[(SIR_{1C}) + (FIR_{1C} \cdot BCF_1)]}$$

where:

SQG_{1C}	=	Soil Quality Guideline for Food and Soil Ingestion for the protection of primary consumers (mg total PCBs·kg^{-1} dry soil);
$DTED_{1C}$	=	daily threshold effects dose for the primary consumer (mg PCBs·kg^{-1} bw_{1C}·d^{-1});
SIR_{1C}	=	soil ingestion rate of the primary consumer (kg dw soil·kg^{-1} bw_{1C}·d^{-1});
FIR_{1C}	=	food ingestion rate of the primary consumer (kg dw food·kg^{-1} bw_{1C}·d^{-1});
BCF_1	=	bioconcentration factor from soil to plant (dw:dw).

Uptake rates of contaminants by plants and invertebrates are often influenced by the contaminant concentrations in the soil (Sample et al. 1997). In the case of PCBs, BCFs are negatively correlated with soil contaminant concentrations as observed by Golder Associates Ltd. and Royal Roads University (1997), who reported a significant negative relationship between soil to plant BCF and total PCB concentrations in soil samples from Resolution Island (BCF = -0.72 $\log_{10}$ soil PCB + 2.089, r^2=0.29, p=0.008). Thus, the data selected to calculate the geometric mean BCF to be used in guideline derivation come from studies in which the reported soil PCB levels are likely to exist in Canada. The geometric mean was the statistic recommended for use by Environment Canada (1994a,b) to obtain a single point estimate BCF from the literature.

Bioconcentration factors for PCBs reported in Appendix V are expressed as the concentration in plant divided by the concentration in soil ([mg·kg^{-1} plant] / [mg·kg^{-1} soil] = kg soil·kg^{-1} plant) and were standardized to obtain kg dry soil·kg^{-1} dry plant values, assuming a ww to dw factor of 4 (US EPA 1993). Studies in which BCF units were unclear or unspecified were excluded from this calculation. The retained values were then used to calculate the geometric mean BCF of 0.14 (dw:dw). This mean value approximately corresponds to the BCFs reported by Strek et al. (1981) and Bacci and Gaggi (1985) for peanut tops and bean leaves, respectively, and will be used in the SQG_{1C} derivation.

Using a value of 0.14 for BCF_1, the SQG_{1C} is calculated as:

SQG_{1C} = (1.00·0.34) / (0.005+ 0.063·0.14) = 24.6 ≈ 25 mg total PCBs·kg^{-1} soil.

Therefore, the Soil Quality Guideline for Soil and Food Ingestion protective of primary consumers is 25 mg total PCBs·kg^{-1} soil. However, since PCBs are known to biomagnify through the food chain, this value should not be considered protective of predators. The derivation of guidelines protective of higher trophic level consumers are presented in the following subsections.

Soil Quality Guideline for Food and Soil Ingestion for the Protection of Secondary Consumers (SQG_{2C})

The second food chain is more complex and involves three trophic levels. It can be represented by either one of the following pathways:

Soil → Plant → Prey (primary consumer) → Predator (secondary consumer)

Soil → Prey (e.g. earthworms) → Predator (secondary consumer)

The model developed to represent this food chain and to derive SQG_{2C} (the subscript 2C stands for secondary consumer) is similar to the one used in deriving SQG_{1C}. However, to account for biomagnification of PCBs from contaminated food and soil to the prey (through the soil→plant→prey or soil→prey pathways), a bioaccumulation factor from soil to prey (BAF_2) will be used, instead of BCF_1.

To be conservative and protective of all secondary consumer species, the food chain pathway involving the prey with the highest soil to prey bioaccumulation factor should be used for guideline derivation. As mentioned in Chapter 3, Terrestrial Plants section of this document, root uptake of PCBs causes little contamination of above-ground foliage which is mainly contaminated through atmospheric deposition, leading to a relatively low soil to plant BCF value of 0.14 (see section Soil Quality Guideline for Food and Soil Ingestion for the Protection of Secondary Consumers (SQG_{2C}) in this chapter). On the other hand, earthworms, which ingest enormous quantities of soil and are in continual contact with it, accumulate considerable amounts of PCBs (Terrestrial Invertebrates section of Chapter 4). Thus it is expected that secondary consumers preying on earthworms will be more at risk than those preying on herbivorous mammals or birds. Given the available literature (Appendices VI and XVI), it was possible to calculate both a soil to small herbivorous mammal BAF and a soil to earthworm BCF. The soil to small herbivorous mammal BAF was calculated using the data for mammals considered mainly herbivorous, including voles (US EPA 1993), field mice (US EPA 1993), squirrels and lemmings. Since bioaccumulation factors used in the derivation of the SQG_{2C} need to be expressed as whole body burdens because predatory birds and mammals often completely consume their prey, and since the available data was mostly in the form of PCB levels in rodent muscles, livers and fat rather than in whole body, whole body BAFs had to be estimated from the organ or tissue values. This was done by multiplying the PCB concentration in the given tissue by the ratio of body weight over tissue weight. These calculations were applied to the data reported by Hendriks et al. (1995) and Greichus and Dohman (1980).

The Watson et al. (1985) study was excluded from these calculations because the measured soil PCB concentrations were 3 to 5 orders of magnitude higher than those in all the other studies used and were therefore considered outliers. An overall geometric mean soil to small herbivorous mammal BAF value of 1.68 was calculated (Appendix XVI). A soil to earthworm geometric mean BCF value of 4.3 was calculated using BCF values from Efroymson et al. 1996, Jones et al. 1996, Kreis et al. 1987, Kihlström et al. 1992, and Belfroid et al. 1995 (Appendix VI). Given these results, a soil to earthworm (primary consumer) to predator (secondary consumer) was chosen as the most appropriate food chain pathway for guideline derivation to protect all secondary consumers.

The shrew was chosen as the best representative secondary consumer since it feeds primarily on earthworms and since toxicological data were available, not directly for shrew, but for a small rodent of identical size, the old field mouse (*Peromyscus polionotus*). Toxicological data indicate that small rodents such as the old field mouse are among the most sensitive species to PCB exposure (Appendix XIV).

The assumptions made in modeling the soil-earthworm-shrew (secondary consumer) pathway are:

- 100% of the contaminant burden in the secondary consumer is accumulated through the ingestion of contaminated food.
- The secondary consumer remains on the contaminated site 100% of the time and preys solely on earthworms.
- 100% of the prey ingested by the secondary consumer are from the PCB contaminated site
- The shrew will be representative of secondary consumers and the dietary concentration causing adverse effects to the old field mouse, a small mammal of identical body weight, will be used for the shrew. Given that the first three assumptions already provide a conservative approach, no additional safety factor was deemed necessary.

The model estimates soil concentrations at a contaminated site that should not be exceeded in order to protect secondary consumer species preying on animals foraging at that site. The model does not consider additional exposure to predators from areas off-site. Since data suggest that such exposure may vary widely by species and geographic region, whether off-site exposure should be factored into final remediation objectives for a site must be considered on a site-specific basis.

A dietary concentration of 5 mg Arochlor 1254·kg^{-1} food was associated with adverse reproductive effects in the old field mouse (*Peromyscus polionotus*) (McCoy et al. 1995). For the shrew (*Blarina brevicauda*), this dietary concentration corresponds to an effects dose of 0.68 mg PCBs·kg^{-1} bw_{2C}·d^{-1} based on a body weight of 0.015 kg (Schlesinger and Potter 1974) and a food consumption rate of 0.009 kg prey ww·d^{-1} (Barrett and Stueck 1976; Buckner 1964).

This effects dose (ED_{2C}) is used to calculate the daily threshold effects dose for the secondary consumer ($DTED_{2C}$) according to the equation:

$$DTED_{2C} = \text{lowest } ED_{2C}/UF$$

where:

$DTED_{2C}$	=	daily threshold effects dose of the secondary consumer (mg PCB·kg^{-1} bw_{2C}·d^{-1});
lowest ED_{2C}	=	lowest observed effects dose (mg PCB·kg^{-1} bw_{2C}·d^{-1});
UF	=	uncertainty factor (if needed); no uncertainty factor was applied since mammalian toxicity data in the literature was adequate.

Therefore,

$DTED_{2C}$ = 0.68 mg total PCBs·kg^{-1} bw_{2C}·d^{-1}

An animal may be exposed to an environmental 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 $DTED_{2C}$. However, since PCB compounds are highly hydrophobic, and therefore have a strong tendency to partition to lipids or organic carbon, it is assumed that 100% (1.00) of exposure can be attributed to the ingestion of food and soil for a terrestrial food chain. It follows then, that exposure from soil and food ingestion should not exceed the $DTED_{2C}$.

$$\text{exposure from direct soil ingestion} + \text{exposure from food ingestion} = 1.00 \cdot DTED_{2C}$$

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. Thus, the secondary consumer's soil ingestion rate is calculated as a proportion of the DMIR according to the equation:

$$SIR_{2C} = DMIR_{2C} \cdot PSI_{2C}$$

where:

SIR_{2C}	=	soil ingestion rate (kg dw soil·kg^{-1} bw_{2C}·d^{-1});
$DMIR_{2C}$	=	dry matter intake rate (kg dry matter·kg^{-1} bw_{2C}·d^{-1});
PSI_{2C}	=	soil ingestion proportions reported with the DMIR.

Given the body weight, the food consumption rate mentioned above, and assuming an 84% water content for earthworms (US EPA 1993), the dry matter intake rate ($DMIR_{2C}$) for the shrew has been calculated to be 0.096 kg dry matter·kg^{-1} bw_{2C}·d^{-1}, with a maximal PSI value of 0.135 associated with soil probing insectivores (McMurter 1993).

$SIR_{2C} = (0.096 \cdot 0.135) = 0.01296$ kg dw soil·kg^{-1} bw_{2C}·d^{-1}

The SIR_{2C} can then be combined with the concentration of the contaminant in the soil (SQG_{2C}) to calculate the exposure from soil ingestion.

$$\text{Exposure from soil ingestion} = SIR_{2C} \cdot SQG_{2C}$$

where:

SIR_{2C}	=	soil ingestion rate of the secondary consumer (kg dw soil·kg^{-1} bw_{2C}·d^{-1});
SQG_{2C}	=	concentration of the contaminant in soil protective of secondary consumers (mg PCB·kg^{-1} soil).

Exposure from food ingestion

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

$$FIR_{2C} = DMIR_{2C} - SIR_{2C}$$

where:

FIR_{2C}	=	food ingestion rate of the secondary consumer (kg dw food·kg^{-1} bw_{2C}·d^{-1});
$DMIR_{2C}$	=	dry matter ingestion rate of the secondary consumer (kg dry matter·kg^{-1} bw_{2C}·d^{-1});
SIR_{2C}	=	soil ingestion rate of the secondary consumer (kg dw soil·kg^{-1} bw_{2C}·d^{-1}).

Thus, $FIR_{2C} = 0.096 - 0.01296 = 0.08304$ kg dw food·kg^{-1} bw_{2C}·d^{-1}

The FIR_{2C} can be combined with the bioaccumulation factor from soil to prey (BAF_2) and the concentration of the contaminant in the soil (SQG_{2C}) to represent the exposure from food ingestion.

$$\text{Exposure from food ingestion} = FIR_{2C} \cdot BAF_2 \cdot SQG_{2C}$$

where:

FIR_{2C} = food ingestion rate of the secondary consumer (kg dw food·kg^{-1} bw_{2C}·d^{-1});
BAF_2 = bioaccumulation factor from soil to prey (dw:dw)
SQG_{2C} = concentration of the contaminant in soil protective of predators (mg PCB·kg^{-1} soil).

Exposure from direct soil ingestion and food ingestion

The equations from soil ingestion and exposure from food ingestion can be combined and rearranged to solve for SQG_{2C}.

$$(SIR_{2C} \cdot SQG_{2C}) + (FIR_{2C} \cdot BAF_2 \cdot SQG_{2C}) = 1.00 \cdot DTED_{2C}$$

Rearranging and solving for SQG_{2C}:

$$SQG_{2C} = \frac{1.00 \cdot DTED_{2C}}{[(SIR_{2C}) + (FIR_{2C} \cdot BAF_2)]}$$

where:

SQG_{2C} = Soil Quality Guideline for Food and Soil Ingestion for the protection of secondary consumers (mg total PCBs·kg^{-1} dry soil);
$DTED_{2C}$ = daily threshold effects dose for secondary consumer (mg PCBs·kg^{-1} bw_{2C}·d^{-1});
SIR_{2C} = soil ingestion rate of secondary consumer (kg dw soil·kg^{-1} bw_{2C}·d^{-1});
FIR_{2C} = food ingestion rate of secondary consumer (kg dw food·kg^{-1} bw_{2C}·d^{-1});
BAF_2 = bioaccumulation factor from soil to prey (dw:dw).

In order to take into consideration the variable behaviour of predator species, this equation must be modified by adding an apportionment factor accounting for the proportion of the foraging range represented by the contaminated site (AF_{FR}) and an apportionment factor accounting for the time spent by the predator on the site (AF_Y). If uncertainty exists regarding these two factors, a value of 1 is recommended. Therefore, the equation becomes:

$$SQG_{2C} = \frac{1.00 \cdot DTED_{2C}}{(SIR_{2C} + FIR_{2C} \cdot BAF_2) \cdot AF_{FR} \cdot AF_Y}$$

where:

AF_{FR} = Proportion of the foraging range which is contaminated;
AF_Y = Proportion of time predator spends on contaminated site.

An important aspect to consider in applying BAFs to estimate biota exposure to contaminants is the potential influence of soil contaminant concentration on organism uptake rates. According to the available literature, the relationship between soil concentrations and bioaccumulation factors from small herbivorous mammals and birds has not been described for PCBs, likely because very little data exist. However, a negative correlation does exist for total PCBs between soil to plant BCFs and soil concentrations (Golder Associates Ltd. and Royal Roads University 1997). Thus, only data from studies in which the reported PCB levels are likely to exist in Canadian soils were used in BAF calculations.

The following values are used for guideline derivation:

$DTED_{2C}$ = 0.68 mg PCB·kg^{-1} bw_{2C}·d^{-1};
SIR_{2C} = 0.01296 kg dw food·kg^{-1} bw_{2C}·d^{-1}
FIR_{2C} = 0.08304 kg dw food·kg^{-1} bw_{2C}·d^{-1};
BAF_2 = 4.3 (dw:dw) (calculated from Appendix VI);
AF_{FR} = 1 (default);
AF_Y = 1 (default).

Thus,

$$SQG_{2C} = \frac{1.00 \cdot 0.68}{(0.01296 + 0.08304 \cdot 4.3) \cdot 1 \cdot 1}$$

SQG_{2C} = 1.8 mg total PCBs·kg^{-1} soil

Therefore, the Soil Quality Guideline for Food and Soil Ingestion protective of secondary consumers is 1.8 mg total PCBs·kg^{-1} soil.

Soil Quality Guideline for Food and Soil Ingestion for the Protection of Tertiary Consumers (SQG_{3C})

The third food chain model, which also involves three trophic levels, is based on the following pathway:

Soil → Invertebrate → Secondary consumer → Predator (tertiary consumer)

According to the available literature, among the terrestrial predators, the most sensitive species to PCBs is the American kestrel *(Falco sparverius),* which exhibited eggshell thinning when fed 3mg·kg^{-1} food of Aroclor 1248 in its diet (Lowe and Stendel 1991). American kestrels will be used as the representative terrestrial predator. For this model, the American kestrel is assumed to prey solely on shrew (*Crocidura russula*, *Sorex araneus*), which are secondary consumers. This assumption is conservative and insures

protection of all tertiairy consumers. Furthermore, earthworms, which are the primary consumers preyed upon by shrew, constitute the base of the food chain and the initial step in contaminant uptake from soils.

The assumptions made in modeling the soil-invertebrate-prey-predator pathway are:

- 100% of the contaminant burden in the predator is accumulated through the ingestion of contaminated food.
- The predator remains on the contaminated site 100% of the time and preys solely on shrew.
- 100% of the prey ingested by the predator are from the PCB contaminated site.
- American kestrels will be representative of the effects of PCBs on raptor species.

Given that the first three assumptions already provide a conservative approach, no additional safety factor was deemed necessary.

For the American kestrel, a dietary concentration of 3 mg PCBs·kg^{-1} prey ww corresponds to an effects dose of 0.87 mg PCBs·kg^{-1} bw_{3C}·d^{-1}, based on a food consumption rate of 0.29 kg prey ww·kg^{-1} bw_{3C}·d^{-1} (U.S. EPA 1993, after Koplin et al. 1980) and a body weight of 0.124 kg (Bloom 1973).

This effects dose (ED_{3C}) is used to calculate the daily threshold effects dose for the secondary consumer ($DTED_{3C}$) according to the equation:

$$DTED_{3C} = \text{lowest } ED_{3C}/UF$$

where:

$DTED_{3C}$	=	daily threshold effects dose of the secondary consumer (mg PCB·kg^{-1} bw_{3C}·d^{-1});
lowest ED_{3C}	=	lowest observed effects dose (mg PCB·kg^{-1} bw_{3C}·d^{-1});
UF	=	uncertainty factor (if needed); no uncertainty factor was applied since avian toxicity data in the literature was adequate.

Therefore, $DTED_{3C}$ = 0.87 mg total PCBs · kg^{-1} bw_{3C} · d^{-1}

As in the previous section, exposure from soil and food ingestion should not exceed the $DTED_{3C}$.

$$\text{Exposure from direct soil ingestion} + \text{exposure from food ingestion} = 1.00 \cdot DTED_{3C}$$

Exposure from direct soil ingestion

The soil ingestion rate (SIR_{3C}) of the American kestrel is considered negligible in this case. While some soil attached to prey may be ingested, the amount is assumed to be

negligible for the American kestrel preying solely on mammals or birds, thus a PSI_{3C} (soil ingestion proportion value of 0 is recommended (see red-tailed hawk in Beyer et al. 1994)). Therefore, predator exposure to PCBs through direct soil ingestion is also negligible (0 kg dw soil·kg^{-1} bw_{3C}·d^{-1}).

Exposure from food ingestion

The food ingestion rate (FIR) of livestock and wildlife is expressed as a portion of the DMIR minus the soil ingestion rate. The FIR is calculated as:

$$FIR_{3C} = DMIR_{3C} - SIR_{3C}$$

where:

FIR_{3C}	=	food ingestion rate of the secondary consumer (kg dw food·kg^{-1} bw_{3C}·d^{-1});
$DMIR_{3C}$	=	dry matter ingestion rate of the secondary consumer (kg dry matter·kg^{-1} bw_{3C}·d^{-1});
SIR_{3C}	=	soil ingestion rate of the secondary consumer (kg dw soil·kg^{-1} bw_{3C}·d^{-1}).

As mentioned previously, the food consumption rate of the American kestrel is 0.29 kg prey ww·kg^{-1} bw_{3C}·d^{-1} (U.S. EPA 1993, after Koplin et al. 1980). This food consumption rate, once converted to dry weight assuming a food water content of 68% in kestrel prey (US EPA 1993), becomes 0.093 kg dw food·kg^{-1} bw_{3C}·d^{-1}.

Thus, FIR_{3C} = 0.093 - 0 = 0.093 kg dw food·kg^{-1} bw_{3C}·d^{-1}

The FIR_{3C} can be combined with the bioaccumulation factor from soil to prey (BAF_3) and the concentration of the contaminant in the soil (SQG_{3C}) to represent the exposure from food ingestion.

$$\text{Exposure from food ingestion} = FIR_{3C} \cdot BAF_3 \cdot SQG_{3C}$$

where:

FIR_{3C}	=	food ingestion rate of the secondary consumer (kg dw food · kg^{-1} bw_{3C} · d^{-1});
BAF_3	=	bioaccumulation factor from soil to prey (dw:dw);
SQG_{3C}	=	concentration of the contaminant in soil protective of predators (mg PCB · kg^{-1} soil).

Exposure from direct soil ingestion and food ingestion

From the previous section:

$$SQG_{3C} = \frac{DTED_{3C}}{(SIR_{3C} + FIR_{3C} \cdot BAF_3) \cdot AF_{FR} \cdot AF_Y}$$

SQG_{3C} = soil quality guideline for ingestion of food and soil protective of tertiary consumers (mg PCB·kg^{-1} soil);
$DTED_{3C}$ = daily threshold effects dose in tertiary consumer food resulting in toxic effects to tertiary consumers (mg PCB·kg^{-1} bw_{3C}·d^{-1});
SIR_{3C} = soil ingestion rate for tertiary consumers (kg dw soil·kg^{-1} bw_{3C}·d^{-1});
FIR_{3C} = food ingestion rate for tertiary consumers (kg dw food·kg^{-1} bw_{3C}·d^{-1});
BAF_3 = bioaccumulation factor from soil to secondary consumer (dw:dw);
AF_{FR} = proportion of the foraging range which is contaminated;
AF_Y = proportion of time tertiary consumer spends on contaminated site.

A soil→worm→shrew geometric mean BAF_3 value of 7.3 (dw:dw) was obtained from Hendriks et al. (1995), based on total PCB measurements from samples collected in the Ochten and Gelderse Poort regions of the Rhine-Delta floodplains. Shrew were thought to feed almost entirely on earthworms. The following values were used in deriving the *SQG*$_{3C}$:

$DTED_{3C}$ = 0.87 mg total PCBs·kg^{-1} bw_{3C}·d^{-1};
FIR_{3C} = 0.093 kg dw food·kg^{-1} bw_{3C}·d^{-1} (US EPA 1993, after Koplin et al. 1980);
BAF_3 = 7.3 (dw:dw, Hendriks et al. 1995);
AF_{FR} = 1 (default);
AF_Y = 1 (default).

Therefore:

$$SQG_{3C} = \frac{1.00 \cdot 0.87}{(0 + 0.093 \cdot 7.3) \cdot 1 \cdot 1}$$

SQG_{3C} = 1.3 mg PCBs·kg^{-1} soil

Therefore, the Soil Quality Guideline for Food and Soil Ingestion protective of tertiary consumers is 1.3 mg PCBs·kg^{-1} soil.

Final Environmental Soil Quality Guidelines

The final environmental soil quality guidelines for PCBs for agricultural, residential/parkland, commercial, and industrial land uses are presented in Table 1. The guidelines are applicable to soils within the pH range of 3.8-7.7, as the toxicological studies upon which these guidelines are based were conducted within this pH range.

Agricultural and residential/parkland land uses

The lowest value from the four procedures used (SQG_{SC}, SQG_{1C}, SQG_{2C}, and SQG_{3C}) is selected as the final environmental soil quality guideline (SQG_E) for residential/parkland and agricultural land uses. The lowest of the four procedures is the SQG_{3C} for third level consumers therefore the CCME recommended environmental soil quality guideline is 1.3 mg PCBs· kg^{-1} soil.

Commercial and industrial land uses

In the case of commercial and industrial land uses, the final environmental soil quality guideline will be based on the direct soil contact guideline (SQG_{SC}). Thus, the CCME recommended soil quality guideline for commercial and industrial land uses is 33 mg PCBs· kg^{-1} soil. However, in site-specific situations where the size and/or location of these land uses may impact higher level consumers, the soil and food ingestion guideline is recommended.

Table 1. Summary of environmental soil quality guidelines for total PCBs

Guidelines	Land Use		
	Agriculture mg PCB·kg^{-1}	Residential/ Parkland mg PCB·kg^{-1}	Commercial/ Industrial mg PCB·kg^{-1}
SQG_{SC}	33	33	33
Nutrient and energy cycle check	—	—	—
SQG_{1C} (Primary consumers)	25	25	—
SQG_{2C} (Secondary consumers)	1.8	1.8	—
SQG_{3C} (Tertiary consumers)	1.3	1.3	—
Groundwater check (aquatic life)	—	—	—
SQG_E	1.3	1.3	33

CHAPTER 6. RECOMMENDED CANADIAN SOIL QUALITY GUIDELINES

According to the formal protocol (CCME 1996), both environmental and human health soil quality guidelines are developed for four land uses: agricultural, residential/parkland, commercial, and industrial. The lowest value generated by the two approaches for each of the four land uses is recommended by the CCME as the Canadian Soil Quality Guideline. The environmental soil quality guidelines for PCBs presented in Chapter 5, were considered along with the human health guidelines (where available; CCME 1999) in making final recommendations for Canadian Soil Quality Guidelines for the protection of environmental and human health. The recommended Canadian Soil Quality Guidelines for the protection of environmental and human health (CCME 1999) are presented below in Table 2. The interim remediation criteria (CCME 1991) are also presented for comparison purposes.

Table 2. Canadian soil quality guidelines for polychlorinated biphenyls (total) (mg· kg^{-1})

	Land use			
	Agricultural	**Residential/ parkland**	**Commercial**	**Industrial**
Guideline	**0.5[a]**	**1.3[b]**	**33[b,c]**	**33[b,c]**
SQG_{HH}	NC[d]	NC[d]	NC[d]	NC[d]
Limiting pathway for SQG_{HH}	ND	ND	ND	ND
Provisional SQG_{HH}	NC[e]	NC[e]	NC[e]	NC[e]
Limiting pathway for provisional SQG_{HH}	ND	ND	ND	ND
SQG_E	1.3	1.3	33	33
Limiting pathway for SQG_E	Soil and food ingestion	Soil and food ingestion	Soil contact	Soil contact
Provisional SQG_E	NC[f]	NC[f]	NC[f]	NC[f]
Limiting pathway for provisional SQG_E	ND	ND	ND	ND
Interim soil quality criterion (CCME 1991)	0.5	5	50	50

Notes: NC = not calculated; ND = not determined; SQG_E = soil quality guideline for environmental health; SQG_{HH} = soil quality guideline for human health.

[a]Data are sufficient and adequate to calculate only an SQE_E, which is greater than the interim soil quality criterion (CCME 1991) for this land use. Therefore, the interim soil quality criterion is retained as the soil quality guideline for this land use.

[b]Data are sufficient and adequate to calculate only an SQG_E, which is less than the existing interim soil quality criterion (CCME 1991) for this land use. Therefore, the SQG_E becomes the soil quality guideline and supersedes the interim soil quality criterion for this land use.

[c]In site-specific situations where the size and/or the location of commercial and industrial land uses may impact higher level consumers, the soil and food ingestion guideline is recommended as the SQG_E.

[d]There is no SQG_{HH} at this time.

[e]There is no provisional SQG_{HH} at this time.

[f]Because data are sufficient and adequate to calculate an SQG_E for this land use, a provisional SQG_E is not calculated.

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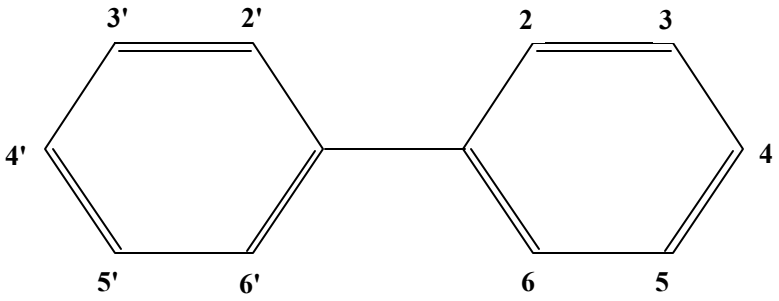
Figure 1. Numbering system of the parent biphenyl molecule of PCBs



Ortho positions:	2, 2', 6, 6'
Meta positions:	3, 3', 5, 5'
Para positions:	4, 4'

Appendix I. IUPAC numbering of PCB congeners

No./	Structure
Monochlorobiphenyls	
1	2
2	3
3	4
Dichlorobiphenyls	
4	2,2'
5	2,3
6	2,3'
7	2,4
8	2,4'
9	2,5
10	2,5'
11	3,3'
12	3,4
13	3,4'
14	3,5
15	4,4'
Trichlorobiphenyls	
16	2,2',3
17	2,2',4
18	2,2',5
19	2,2',6
20	2,3,3'
21	2,3,4
22	2,3,4'
23	2,3,5
24	2,3,6
25	2,3',4
26	2,3',5
27	2,3',6
***28**	2,4,4'
29	2,4,5
30	2,4,6
31	2,4',5
32	2,4',6
33	2',3,4
34	2',3,5
35	3,3',4
36	3,3',5
****37**	3,4,4'
38	3,4,5
39	3,4',5

No./	Structure
Tetrachlorobiphenyls	
40	2,2',3,3'
41	2,2',3,4
42	2,2',3,4'
43	2,2',3,5
44	2,2',3,5'
45	2,2',3,6
46	2,2',3,6'
47	2,2',4,4'
48	2,2',4,5
49	2,2',4,5'
50	2,2',4,6
51	2,2',4,6'
52	2,2',5,5'
53	2,2',5,6'
54	2,2',6,6'
55	2,3,3',4
56	2,3,3',4'
57	2,3,3',5
58	2,3,3',5'
59	2,3,3',6,
***60**	2,3,4,4'
61	2,3,4,5
62	2,3,4,6
63	2,3,4',5
64	2,3,4',6
65	2,3,5,6
66	2,3',4,4'
67	2,3',4,5
68	2,3',4,5'
69	2,3',4,6
70	2,3',4',5
71	2,3',4',6
72	2,3',5,5'
73	2,3',5',6
74	2,4,4',5
75	2,4,4',6
76	2',3,4,5
****77**	3,3',4,4'
78	3,3',4,5
79	3,3',4,5'
80	3,3',5,5'
****81**	3,4,4',5

No./	Structure
Pentachlorobiphenyls	
82	2,2',3,3',4
83	2,2',3,3',5
84	2,2',3,3',6
85	2,2'3,4,4'
86	2,2',3,4,5
87	2,2',3,4,5'
88	2,2',3,4,6
89	2,2',3,4,6'
90	2,2',3,4',5
91	2,2',3,4',6
92	2,2',3,5,5'
93	2,2',3,5,6
94	2,2',3,5,6'
95	2,2',3,5',6
96	2,2',3,6,6'
97	2,2',3',4,5
98	2,2',3',4,6
99	2,2',4,4',5
100	2,2',4,4',6
101	2,2',4,5,5'
102	2,2',4,5,6'
103	2,2',4,5',6
104	2,2',4,6,6'
***105**	2,3,3',4,4'
106	2,3,3',4,5
107	2,3,3',4',5
108	2,3,3',4,5'
109	2,3,3',4,6
110	2,3,3',4',6
111	2,3,3',5,5'
112	2,3,3',5,6
113	2,3,3',5',6
***114**	2,3,4,4',5
115	2,3,4,4',6
116	2,3,4,5,6
117	2,3,4',5,6
***118**	2,3',4,4',5
119	2,3',4,4',6
120	2,3',4,5,5'
121	2,3',4,5',6
122	2',3,3',4,5
***123**	2',3,4,4',5
124	2',3,4,5,5'
125	2',3,4,5,6'
****126**	3,3',4,4',5
127	3,3',4,5,5'

No./	Structure
Hexachlorobiphenyls	
128	2,2',3,3',4,4'
129	2,2',3,3',4,5
130	2,2',3,3',4,5'
131	2,2',3,3',4,6
132	2,2',3,3',4,6'
133	2,2',3,3',5,5'
134	2,2',3,3',5,6
135	2,2',3,3',5,6'
136	2,2',3,3',6,6'
137	2,2',3,4,4',5
138	2,2',3,4,4',5'
139	2,2',3,4,4',6
140	2,2',3,4,4',6'
141	2,2',3,4,5,5'
142	2,2',3,4,5,6
143	2,2',3,4,5,6'
144	2,2',3,4,5',6
145	2,2',3,4,6,6'
146	2,2',3,4',5,5'
147	2,2',3,4',5,6
148	2,2',3,4',5,6'
149	2,2',3,4',5',6
150	2,2',3,4',6,6'
151	2,2',3,5,5',6
152	2,2',3,5,6,6'
153	2,2',4,4',5,5'
154	2,2',4,4',5,6'
155	2,2',4,4',6,6'
***156**	2,3,3',4,4',5
***157**	2,3,3',4,4',5'
158	2,3,3',4,4',6
159	2,3,3',4,5,5'
160	2,3,3',4,5,6
161	2,3,3',4,5',6
162	2,3,3',4',5,5'
163	2,3,3',4',5,6
164	2,3,3',4',5',6
165	2,3,3',5,5',6
166	2,3,4,4',5,6
***167**	2,3',4,4',5,5'
168	2,3',4,4',5',6
****169**	3,3',4,4',5,5'

No./	Structure
Heptachlorobiphenyls	
170	2,2',3,3',4,4',5
171	2,2',3,3',4,4',6
172	2,2',3,3',4,4',5'
173	2,2',3,3',4,5,6
174	2,2',3,3',4,5,6'
175	2,2',3,3',4,5',6
176	2,2',3,3',4,6,6'
177	2,2',3,3',4',5,6
178	2,2',3,3',5,5',6
179	2,2',3,3',5,6,6'
180	2,2',3,4,4',5,5'
181	2,2',3,4,4',5,6
182	2,2',3,4,4',5,6'
183	2,2',3,4,4',5',6
184	2,2',3,4,4',6,6'
185	2,2',3,4,5,5',6
186	2,2',3,4,5,6,6'
187	2,2',3,4',5,5',6
188	2,2',3,4',5,6,6'
***189**	2,3,3',4,4',5,5'
190	2,3,3',4,4',5,6
191	2,3,3',4,4',5',6
192	2,3,3',4,5,5',6
193	2,3,3',4',5,5',6
Octachlorobiphenyls	
194	2,2',3,3',4,4',5,5'
195	2,2',3,3',4,4',5,6
196	2,2',3,3',4,4',5',6
197	2,2',3,3',4,4',6,6'
198	2,2',3,3',4,5,5',6
199	2,2',3,3',4,5,6,6'
200	2,2',3,3',4,5',6,6'
201	2,2',3,3',4',5,5',6
202	2,2',3,3',5,5',6,6'
203	2,2',3,4,4',5,5',6
204	2,2',3,4,4',5,6,6'
205	2,3,3',4,4',5,5',6
Nonachlorobiphenyls	
206	2,2',3,3',4,4',5,5',6
207	2,2',3,3',4,4',5,6,6'
208	2,2',3,3',4,5,5',6,6'
Decachlorobiphenyl	
209	2,2',3,3',4,4',5,5',6,6'

Source: Ballschmiter and Zell 1980

(*) mono- and (**) non-*ortho* substituted coplanar congeners potentially detected in environmental samples (Metcalfe and Haffner 1995).

Appendix II. Chemical and physical properties of groups of PCB congeners

Congener Group	Empirical Formulae	# of Possible Congeners	% Cl by Weight	Molecular Weight ($g \cdot mol^{-1}$)	Aqueous Solubility* ($\mu g \cdot L^{-1}$)	Vapour Pressure* (Pa)	Henry's Law Constant* ($Pa \cdot m^3 \cdot mol^{-1}$)	Log K_{ow}*
Biphenyl	$C_{12}H_{10}$	1	0.0	154.2	7000	2.43	53.5	3.9
Mono-CBP	$C_{12}H_9Cl$	3	18.8	187.6	1200-5500	0.271-2.04	42.56-75.55	4.3-4.6
Di-CBP	$C_{12}H_8Cl_2$	12	31.8	223.1	60-2000	0.0018-0.279	17.0-92.2	4.9-5.3
Tri-CBP	$C_{12}H_7Cl_3$	24	41.4	257.5	15-400	0.0136-0.143	24.3-92.2	5.5-5.9
Tetra-CBP	$C_{12}H_6Cl_4$	42	48.6	292.0	4.3-100	0.000059-0.0054	1.72-47.6	5.6-6.5
Penta-CBP	$C_{12}H_5Cl_5$	46	54.4	326.4	4-20	0.000304-0.0093	24.8-151.4	6.2-6.5
Hexa-CBP	$C_{12}H_4Cl_6$	42	58.9	360.9	0.4-1.0	0.000020-0.00159	11.9-81.8	6.7-7.3
Hepta-CBP	$C_{12}H_3Cl_7$	24	62.8	395.3	0.45-2.0	0.0000273	5.4	6.7-7.0
Octa-CBP	$C_{12}H_2Cl_8$	12	66.0	429.8	0.2-0.3	0.0000266	38.1	7.1
Nona-CBP	$C_{12}HCl_9$	3	68.7	464.2	0.018-0.11	0.0000063	NA	7.2-8.16
Deca-CBP	$C_{12}Cl_{10}$	1	71.2	498.7	0.0012	0.00000005	20.84	8.26

Source: Mackay et al. 1992 (values reported were selected from Shiu and Mackay 1986, Shiu et al. 1987, and Mackay et al. 1986)
CBP = chlorobiphenyl, NA = not available
* ranges represent differences among congeners within PCB groups

Appendix III. Existing Environmental Quality Criteria and Guidelines for PCBs in Soil

Jurisdiction	Guideline ($mg \times kg^{-1}$ dw)	PCB Formulation	Category	Reference
Alberta	0.5	total PCBs	Assessment and remediation criterion	Alberta Environment 1994
Ontario	0.5 5 25 25	total PCBs total PCBs total PCBs total PCBs	Surface soil clean-up criteria - Agricultural land use Surface soil clean-up criteria - Residential/Parkland land use Surface soil clean-up criteria - Industrial/Commercial Sub-surface soil clean-up criteria - Residential/Parkland	OMEE 1994
Quebec	A: <0.1 B: 1 C: 10	Aroclors 1242, 1248, 1254 and 1260	Natural background level Investigation criterion Remediation criterion	MENVIQ 1990
Saskatchewan	5		Target level for cleanup	Strachan 1988
France	0.05 1 5 10	total PCBs	Background maximum Investigation threshold Treatment threshold Emergency (immediate action) threshold	ASTM 1995
Netherlands	0.02 1 0.001 0.004	∑6 PCBs (PCBs 28, 52, 101, 138, 153, 180) ∑7 PCBs (PCBs 28, 52, 101, 118, 138, 153, 180) PCBs 28 and 52 (individual) PCBs 101, 138, 153, 180 (individual)	Standard soil target value Standard soil intervention value Standard soil target value Standard soil target value	MHSPE 1994
Netherlands	4.8 0.2 ≥10 0.5 0.1 0.2 0.1-0.5	PCB 28 PCB 52 PCB 28 PCB 52 PCB 138/153 PCB 138/153 PCB 138/153	Agricultural threshold level - Grassland Agricultural threshold level - Grassland Agricultural threshold level - Arable land, fodder production Agricultural threshold level - Arable land, fodder production Agricultural threshold level - Grassland Agricultural threshold level - Arable land, fodder production Agricultural threshold level - Arable land, food production	MHSPE 1994
Sweden 4 7	 0.02 	 total PCBs total PCBs 	Generic guideline values for contaminated soils: Land with sensitive use Land with less sensitive use and groundwater extraction Land with less sensitive use	SEPA 1997
United States 25 50 10		total PCBs	EPA contaminated soil cleanup policy: Restricted areas Electrical substation restricted areas Non-restricted areas	ASTM 1995
United States	0.09	total PCBs	Action level - soil	ASTM 1995
United States	0.92 0.92 0.92 0.92 0.92 1.8 1.8 10	Aroclor 1016 Aroclor 1221 Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254 Aroclor 1260 total PCBs	Non-wastewater universal treatment standards	ASTM 1995
Arizona	0.18	total PCBs	Ingestion health-based guidance level	ASTM 1995

Appendix III (continued). Existing Environmental Quality Criteria and Guidelines for PCBs in Soil

Jurisdiction	Guideline ($mg \cdot kg^{-1}$ dw)	PCB Formulation	Category	Reference
Hawaii	1.0	total PCBs	Soil - Drinking water source	ASTM 1995
	1.0	total PCBs	Soil - Non-drinking water source	
Massachusetts	2	total PCBs	S-1 soil and GW-1, GW-2, GW-3	ASTM 1995
	2	total PCBs	S-2 soil and GW-1, GW-2, GW-3	
	2	total PCBs	S-3 soil and GW-1, GW-2, GW-3	
Michigan	1.0	total PCBs	Direct contact value	ASTM 1995
New Jersey	0.49	total PCBs	Residential - Direct contact	ASTM 1995
	2.0	total PCBs	Non-residential - Direct contact	
	100	total PCBs	Soil - Impact to groundwater	
New York	1.0	total PCBs	Surface soil guideline	ASTM 1995
	10	total PCBs	Subsurface soil guideline	
Oregon	0.08	total PCBs	Soil regulation	ASTM 1995
	0.7	total PCBs	Industrial sites	
	0.002 $mg \cdot L^{-1}$	total PCBs	Soil leachate reference concentration	
Tennessee	0.08	total PCBs	Required detection level	ASTM 1995
	10	total PCBs	Guideline - Industrial area	
	1	total PCBs	Guideline - Residential area	
Texas	0.05	total PCBs	Residential - Protective of groundwater	ASTM 1995
	0.05	total PCBs	Industrial - Protective of groundwater	
	10	total PCBs	Residential - Protective of human health	
	25	total PCBs	Industrial - Protective of human health	
Washington	10	total PCBs	Industrial area	ASTM 1995
	1	total PCBs	Residential area	
Wyoming	0.2	total PCBs	Soil - oral ingestion	ASTM 1995

Appendix IV. Existing Environmental Quality Criteria and Guidelines for PCBs in Groundwater

Jurisdiction	Guideline ($mg \cdot L^{-1}$)	PCB Formulation	Category	Reference
Ontario	0.0002	total PCBs	Groundwater clean-up criteria - All land use categories	OMEE 1994
Quebec	A: <0.0001 B: 0.0002 C: 0.001	Aroclors 1242, 1248, 1254 and 1260	Natural background level Investigation criterion Remediation criterion	MENVIQ 1990
Netherlands	0.00001 0.00001 0.00001	total PCBs ∑6 PCBs (PCBs 28, 52, 101, 138, 153, 180) ∑7 PCBs (PCBs 28, 52, 101, 118, 138, 153, 180)	Target value Target value Intervention value	MHSPE 1994
United States	$5x10^{-6}$	total PCBs	Action level	ASTM 1995
United States	0.0005 0	total PCBs total PCBs	Groundwater maximum contaminant level Groundwater maximum contaminant level goal	ASTM 1995
Alaska	0.01	Decachlorobiphenyls	Groundwater maximum contaminant level	ASTM 1995
Arizona	$5x10^{-6}$	total PCBs	Ingestion health-based guidance level	ASTM 1995
Colorado	$7.9x10^{-8}$ $5x10^{-6}$	total PCBs total PCBs	Groundwater standard - Rocky Flat Aquifer Elsewhere	ASTM 1995
Connecticut	$4x10^{-6}$	total PCBs	Groundwater risk-based criteria	ASTM 1995
Hawaii	0.0005 0.01	total PCBs total PCBs	Groundwater - Drinking water source Groundwater - Non-drinking water source	ASTM 1995
Illinois	0.005 0.025	total PCBs Decachlorobiphenyls	Groundwater - Class I Groundwater - Class II	ASTM 1995
Kansas	$5x10^{-6}$ $1.4x10^{-5}$ $5x10^{-5}$ 0.002	total PCBs total PCBs total PCBs total PCBs	Kansas Notification Level Alternate Kansas Notification Level Kansas Action Level Alternate Kansas Action Level	ASTM 1995
Massachusetts	0.0005 0.0003	total PCBs total PCBs	GW-1 standard GW-3 standard	ASTM 1995
Michigan	$1.8x10^{-5}$ $2x10^{-8}$	total PCBs total PCBs	Health-based drinking water value Groundwater-surface water interface values	ASTM 1995
Nebraska	0.0005	total PCBs	Groundwater regulation	ASTM 1995
New Jersey	0.0005	total PCBs	Groundwater - Class IIA	ASTM 1995
New York	0.0001	total PCBs	Groundwater guideline	ASTM 1995
Oregon	$1x10^{-5}$	total PCBs	Groundwater reference concentration	ASTM 1995
Rhode Island	0.0005 0.00025	total PCBs total PCBs	Groundwater quality standard Groundwater preventative action limit	ASTM 1995
South Dakota	$5x10^{-5}$	total PCBs	Groundwater regulation	ASTM 1995
Tennessee	0.0005	total PCBs	Guideline	ASTM 1995
Texas	0.0005	total PCBs	Groundwater regulation	ASTM 1995
Vermont	$8x10^{-6}$ $8x10^{-7}$	total PCBs total PCBs	Enforcement standard Preventative action limit	ASTM 1995
Washington	0.0001	total PCBs	Groundwater regulation	ASTM 1995
Wisconsin	$3x10^{-5}$ $3x10^{-6}$	total PCBs total PCBs	Proposed enforcement standard Proposed preventative action limit	ASTM 1995
Wyoming	0.0005	total PCBs	Groundwater regulation	ASTM 1995

Appendix V. Soil-to-Plant Bioconcentration Factors

Species	"Organ"	PCB Formulation	Soil Type	Exposure Duration	Exposure Level ($mg \times kg^{-1}$)	Bioconcentration Factor	Reference
Terrestrial Plants							
Beet (*Beta vulgaris*)	plant tops	Aroclor 1254		39 days	20	0.041	Strek et al. 1981
Sugar beet	whole plant root peels peeled roots	Aroclor 1254	brown soil		0.3	0.01 - 0.5 0.17 0.03	Wallnöfer et al. 1975 cited in Pal et al. 1980
Sugar beet	roots leaves	PCBs			0.24 (0-10 cm) 0.17 (10-20 cm)	0.07 0.03	Moza et al. 1976 cited in Pal et al. 1980
Weeds (species unknown)	n.r.*	PCBs			0.24 (0-10 cm) 0.17 (10-20 cm)	0.80	Moza et al. 1976 cited in Pal et al. 1980
Weeds (*Panicum virgatum, Panicum bisulcatum*)	whole plant	^{14}C-labelled Aroclor 1254	Lakeland sand	n.r.*	100	<0.01	Strek and Weber 1980
Sorghum (*Sorghum bicolor)*)	plant tops	Aroclor 1254		39 days	20	0.003	Strek et al. 1981
Peanut (*Arachis hypogaea*)	plant tops	Aroclor 1254		78 days	20	0.024	Strek et al. 1981
Soybean	plant tops	Aroclor 1254	Lakeland sand	16 days	0-1000	0.016	Weber and Mrozek 1979
Soybean sprouts	stems and tops	Aroclor 1242 Aroclor 1254	sand	10 days 15 days	100 100	0.0015	Suzuki et al. 1977
Fescue	plant tops	Aroclor 1254	Lakeland sand	50 days	0-1000	0.17	Weber and Mrozek 1979
Tomato	n.r.*	several congeners	soil and vermiculite	until maturation	n.r.*	0	Pal et al. 1980
Corn	leaves	Aroclor 1254 Aroclor 1260	soil-sludge system	n.r.*	92-144 µg PCBs·L^{-1} sludge	< 1	Lawrence and Tosine 1977
Corn (*Zea mays*)	plant tops	Aroclor 1254		13 days	20	0.001	Strek et al. 1981
Corn	grain stover	sewage sludge	Plano silt-loam	season	various	0 0	Gan and Berthouex 1994
Broad bean	leaves	Fenclor 64	Pliocene sand, 0.15% o.m.	28 days	460	0.06	Bacci and Gaggi 1985
Bean	leaves	Fenclor 64	Pliocene sand, 0.15% o.m.	28 days	460	0.09	Bacci and Gaggi 1985
Cucumber	leaves	Fenclor 64	Pliocene sand, 0.15% o.m.	28 days	460	0.05	Bacci and Gaggi 1985
Tomato	leaves roots	Fenclor 64	Pliocene sand, 0.15% o.m.	28 days	460	0.03 0.3	Bacci and Gaggi 1985
Radish	n.r.* n.r.* peels n.r.*	Aroclor 1254	acid soil acid soil brown sand acid soil	57 days? 57 days? 57 days? 68 days	0.05 0.5 0.2 5	0 0 0.02 0.005	Wallnöfer et al. 1975 cited in Pal et al. 1980
Carrot	roots	Aroclor 1254			100	0.16	Iwata et al. 1974 cited in Pal et al. 1980

Appendix V (continued). Soil-to-Plant Bioconcentration Factors

Species	"Organ"	PCB Formulation	Soil Type	Exposure Duration	Exposure Level ($mg \times kg^{-1}$)	Bioconcentration Factor	Reference
Carrot	n.r.* n.r.* n.r.* roots roots	Aroclor 1254	acid soil acid soil acid soil acid soil brown sand	120 days? 120 days? 120 days? 120 days 81 days	0.05 0.5 5 5 0.5	0 << 1 < 1 0.16 0.16	Wallnöfer et al. 1975 cited in Pal et al. 1980
Carrot	roots leaves	DiCB		112 days	0.118	2 0.92	Moza et al. 1976, cited in WHO 1993
Carrot	roots	2,4',5-trichlorobiphenyl	52.2% sand, 34.5% silt, 13.3% clay, 0.3% o.m.	4-6 months	0.313	2.7	Moza et al. 1979
Carrot	top root peel top root peel	Aroclor 1248	Bluepoint sandy loam Glendale clay loam	75 days	0.75 0.75	<0.03 <0.03 3.00 <0.03 <0.03 0.07	O'Connor et al. 1990

* n.r. = not reported

Appendix VI. Soil-to-Earthworm Bioconcentration Factors

Species	Depurated	PCB Formulation	Exposure duration	Worm concentration ($mg \cdot kg^{-1}$)	Soil concentration ($mg \cdot kg^{-1}$)	Bioconcentration Factor (dw:dw)	Reference
Various	yes	total PCBs	resident	3588 2063 3219 95 59 87.5	55 139 137 12.5 8 5.5	65.2 14.8 23.5 7.6 7.3 15.9	Diercxsens et al. 1985
Nicodrilus sp.	yes	total PCBs	resident	174 57 55 11 22 31 13 61 76 19	13 5 4 1 3 3 3 6 6 3	13.4 11.4 5.5 11 7.3 10.3 4.3 10.1 12.7 6.3	Kreis et al. 1987
Various	yes	Aroclor 1254	resident	0.07 0.21 0.08	0.11 0.34 0.5	0.64 0.62 0.16	Efroymson et al. 1996 and Jones et al. 1996
		Aroclor 1260		0.07 0.07 0.91 0.05 0.17 0.09 0.3 0.06 6.3 0 1.1 1.1 0.21	0.01 0.04 0.07 0.08 0.08 0.1 0.14 0.15 0.28 0.3 0.33 0.42 0.5	7 1.75 13 0.63 2.13 0.9 2.14 0.4 22.5 0 3.33 2.62 0.42	
Eissenia andrei	yes	$\sum$7 PCBs	60 d	2.25	0.68	0.31	Belfroid et al. 1995

Appendix VII. Selected Microbial Processes Studies

Organism	Effect	Endpoint (% reduction)	Concentration	PCB Formulation	Soil pH	Soil Condition	Reference
Soil microbes	Nitrification	EC (n.s.)	14.0 $ng \cdot g^{-1}$	PCBs	7.7	24.2% clay, 40-60 mg $P \cdot kg^{-1}$, 120-140 mg $Mg \cdot kg^{-1}$, 300-350 mg $K \cdot kg^{-1}$	Dusek 1995
Soil microbes	Biomass content Respiration	EC (23%) EC (14%)	14.0 $ng \cdot g^{-1}$	PCBs	7.7	24.2% clay, 40-60 mg $P \cdot kg^{-1}$, 120-140 mg $Mg \cdot kg^{-1}$, 300-350 mg $K \cdot kg^{-1}$	Dusek and Tesarova 1996

Appendix VIII. Selected Plant Aroclor 1254 Toxicity Studies

Organism	Effect	Endpoint (% reduction)	Concentration ($mg \times kg^{-1}$)	PCB Formulation	Soil pH	Soil Condition	Reference
Fescue (*Fescue arundinacea*)	42 days: Height Fresh top weight	 EC (0%) EC (16%)	 1000 1000	Aroclor 1254	4.7	Lakeland sand A horizon, 1% organic matter, 5% clay, 6% silt	Weber and Mrozek 1979
Soybeans (*Glycine max*)	26 days: Height Fresh top weight	 EC (15%) EC (22%)	 1000 1000	Aroclor 1254	4.7	Lakeland sand A horizon, 1% organic matter, 5% clay, 6% silt	Weber and Mrozek 1979
Soybeans (*Glycine max*)	2nd crop 23 days: Height Fresh top weight Water use 3rd crop 24 days: Height Fresh top weight Water use	 EC (22%) EC (24%) EC (47%) EC (20%) EC (37%) EC (32%)	 1000 1000 1000 1 1000 1000	Aroclor 1254	6.0	Lakeland sand A horizon, CEC=1.5 mequiv/100g, 1% organic matter, 5% clay, 6% silt	Strek et al. 1981
Beet (*Beta vulgaris*)	14 days: Height 28 days: Height Fresh top weight 56 days: Height Fresh top weight 32-53 days Water use	 EC (63%) EC (73%) EC (100%) EC (100%) EC (100%) EC (96%)	 1000 1000 1000 1000 1000 1000	Aroclor 1254	6.0	Lakeland sand A horizon, CEC=1.5 mequiv/100g, 1% organic matter, 5% clay, 6% silt	Strek et al. 1981
Corn (*Zea mays*)	5 days: Height 18 days: Height Fresh top weight Water use	 EC (30%) EC (0%) EC (0%) EC (0%)	 100 1000 1000 1000	Aroclor 1254	6.0	Lakeland sand A horizon, CEC=1.5 mequiv/100g, 1% organic matter, 5% clay, 6% silt	Strek et al. 1981
Sorghum (*Sorghum bicolor*)	69 days: Height Fresh top weight Water use	 EC (0%) EC (0%) EC (0%)	 1000 1000 1000	Aroclor 1254	6.0	Lakeland sand A horizon, CEC=1.5 mequiv/100g, 1% organic matter, 5% clay, 6% silt	Strek et al. 1981

Appendix VIII (continued). Selected Plant Aroclor 1254 Toxicity Studies

Organism	Effect	Endpoint (% reduction)	Concentration ($mg \times kg^{-1}$)	PCB Formulation	Soil pH	Soil Condition	Reference
Panicum sp. (*Panicum virgatum, Panicum bisulcatum*)	Plant height Plant weight	EC (0%) EC (0%)	100 100	Labelled Aroclor 1254	6.0	Lakeland sand A horizon, CEC=1.5 mequiv/100g, 1% organic matter, 5% clay, 6% silt	Strek and Weber 1980
Redroot pigweed (*Amaranthus retroflexus*)	28 days: Height	EC (31%)	150	Aroclor 1254	6.0	Lakeland sand A horizon, CEC=1.5 mequiv/100g, 1% organic matter, 5% clay, 6% silt	Strek and Weber 1982

Appendix IX. Selected Invertebrate Aroclor 1254 Toxicity Studies

Organism	Effect	Endpoint (% reduction)	Concentration ($mg \times kg^{-1}$)	PCB Formulation	Soil pH	Soil Condition	Reference
House crickets (*Acheta domesticus*)	Mortality	LC_{50}	1200	Aroclor 1254	5.5	Hosmer silt loam, 4% organic matter	Paine et al. 1993
Soil fauna	Nematodes:			Aroclor 1254	3.8	33% sand, 56% silt, 11% clay, CEC=6.2 mequiv/100g	Parmelee et al. 1997
	Total numbers	EC (0%)	2500				
	Trophic group numbers	EC (0%)	2500				
	Microarthropods:						
	Total numbers	EC (87%)	2500				
	Trophic group numbers	EC (n.s.)	2500				

Appendix X. Consulted Microbial Processes Studies

Organism	Effect	Endpoint (% reduction)	Concentration	PCB Formulation	Soil pH	Soil Condition	Reference
Soil alga (*Navicula pellicosa*)	Photosynthesis Cell numbers	EC (51-72%) EC (46-66%)	20 $\mu g \cdot L^{-1}$	Aroclor 1221, 1242, 1016, 1248	N/A	N/A (liquid culture)	Glooschenko and Glooschenko 1975
Soil microfungi (*Aspergillus flavus*)	Mycelial growth RNA content DNA content	EC (1.4-55%)	5, 10, 25, 50 $mg \cdot kg^{-1}$	Aroclor 1232, 1242, 1248, 1254, 1260	N/A	N/A (liquid culture)	Murado et al. 1976
Amoeba (*Acanthamoeba* sp.)	Mortality Predation	EC	2500 $mg \cdot kg^{-1}$	Pyralene	6.2	Silt loam - 31.4% clay, 36.4% silt, 32.2% sand, 2.64% organic matter	Steinberg et al. 1990

Appendix XI. Consulted Plant Toxicity Studies

Organism	Effect	Endpoint (% reduction)	Concentration	PCB Formulation	Soil pH	Soil Condition	Reference
Ostrich fern (*Matteuccia struthiopteris*)	Gametophytic mutations	5.2X higher frequency of mutations	26.3 $mg \cdot kg^{-1}$	PCBs	NR	Flood plain of Housatonic River (field study)	Klekowski 1982
Saltmarsh vegetation (*Spartina alterniflora* Loisel)	In sand: Height # live leaves/stem Stems/plant In mud: Stems/plant	 EC (30%) EC (25%) EC (-300%) EC (75%)	 30.7 $\mu g \cdot kg^{-1}$ 30.7 $\mu g \cdot kg^{-1}$ 30.8 $\mu g \cdot kg^{-1}$ 2.2 $\mu g \cdot kg^{-1}$	PCBs (similar to Aroclor 1254)	NR	NR	Mrozek et al. 1983
Prairie grasses: Smooth brome (*Bromus inermis*) Streambank wheatgrass (*Agropyron riparium*) Crested wheatgrass (*Agropyron cristatum*) Orchard grass (*Dactylis glomerata*)	Seed germination	EC (90%) EC (90%) EC (90%) EC (94%)	130 $\mu g \cdot kg^{-1}$ 110 $\mu g \cdot kg^{-1}$ 130 $\mu g \cdot kg^{-1}$ 33 $\mu g \cdot kg^{-1}$	Aroclor 1260	8.6	Parking lot “soil”: Sandy loam, 0.5% organic matter, 9 mg NO_3-N· kg^{-1}, 6 mg P· kg^{-1}	Siciliano et al. 1997

Appendix XII. Consulted Invertebrate Toxicity Studies

Organism	Effect	Endpoint (% reduction)	Concentration	PCB Formulation	Soil pH	Soil Condition	Reference
Earthworms (*Eisenia veneta*)	Immune function	EC (65%)	10 $\mu g \cdot cm^{-2}$ (filter paper)	Aroclor 1254	N/A	N/A	Bunn et al. 1996
Eathworms (*Eisenia foetida*)	Mortality	LC_{50} LD_{50}	30.4 $\mu g \cdot cm^{-2}$ (filter paper) 4500 $\mu g \cdot g^{-1}$	Aroclor 1254	N/A	N/A	Fitzpatrick et al. 1992
Snails (*Cepeae nemoralis*)	Rasping behaviour: Shell damage Shell penetration	 EC(-16%) EC(-7%)	 0.5 $mg \cdot kg^{-1}$ 5.0 $mg \cdot kg^{-1}$	Aroclor 1016, 1254	N/A	N/A	Hatch and Allen 1979
Bacteriophagus nematode (*Acrobeloides nanus*)	5 days: # Individuals # Eggs Nematode weight 10 days: # Individuals # Eggs Nematode weight	 EC (0%) EC (0%) EC (0%) EC (38%) EC (47%) EC (50%)	 60 $\mu g \cdot dish^{-1}$ 60 $\mu g \cdot dish^{-1}$ 60 $\mu g \cdot dish^{-1}$ 15 $\mu g \cdot dish^{-1}$ 15 $\mu g \cdot dish^{-1}$ 15 $\mu g \cdot dish^{-1}$	Aroclor 1254	N/A	N/A (Study was conducted in petri dishes)	Wasilewska et al. 1975

Appendix XIII. Avian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet ($mg \times kg^{-1}$)	Route of Exposure	Exposure Period	Estimated Dose** ($mg \times kg^{-1} bw \cdot d^{-1}$)	Reference
Primary Consumers:							
Bobwhite quail *(Colinus virginanus)*	Aroclor 1221	LC_{50}	>6000	diet	5 days	958	Hill et al. 1975
	Aroclor 1232	LC_{50}	3002			480	
bw = 0.072 kg (21 d old, Shellenberger 1978)	Aroclor 1242	LC_{50}	2098			335	
fc = 0.0115 $kg \cdot d^{-1}$ (Hill and Camardese 1986)	Aroclor 1248	LC_{50}	1175			188	
	Aroclor 1254	LC_{50}	604			96	
	Aroclor 1260	LC_{50}	747			119	
	Aroclor 1262	LC_{50}	871			139	
Japanese quail *(Coturnix coturnix japonica)*	Aroclor 1221	LC_{50}	>5000	diet	5 days	>799	Hill and Camardese 1986
	Aroclor 1232	LC_{50}	>5000			>799	
bw = 0.072 kg (21 d old, Shellenberger 1978)	Aroclor 1242	LC_{50}	>5000			>799	
fc = 0.0115 $kg \cdot d^{-1}$ (Hill and Camardese 1986)	Aroclor 1248	LC_{50}	4819			770	
	Aroclor 1254	LC_{50}	2929			468	
	Aroclor 1260	LC_{50}	2195			351	
	Aroclor 1262	LC_{50}	2304			368	
Japanese quail *(Coturnix coturnix japonica)*	Clophen A60	eggshell breaking strength:		diet	3 weeks		Biessman 1982
bw = 0.072 kg (21 d old, Shellenberger 1978)		EC (8%)	50			7.3	
fc = 0.0105 $kg \cdot d^{-1}$ (Nagy 1987)		liver weight:					
		EC (20%)	50			7.3	
Japanese quail *(Coturnix coturnix japonica)*		egg production:		diet	3 weeks		Call and Harrell 1974
	Aroclor 1242	EC (18%)	312.5			35.2	
bw = 0.15 kg (Vos et al. 1971)	Aroclor 1254	EC (47%)	78.1			8.8	
fc = 0.0169 $kg \cdot d^{-1}$ (Nagy 1987)	Aroclor 1260	EC (13%)	62.5			7.0	
		shell thickness:					
	Aroclor 1242	EC (5%)	312.5			35.2	
	Aroclor 1254	EC (12%)	78.1			8.8	
	Aroclor 1260	EC (10%)	62.5			7.0	
Japanese quail *(Coturnix coturnix japonica)*	Aroclor 1248	egg production, hatchability:	20	diet	8 weeks	2.3	Scott et al. 1975
bw = 0.15 kg (Vos et al. 1971)		EC (0%)					
fc = 0.0169 $kg \cdot d^{-1}$ (Nagy 1987)							
Japanese quail *(Coturnix coturnix japonica)*	Aroclor 1254	avoidance response:		diet	8 days		Kreitzer and Heinz 1974
		EC (10-15%)	200			29.2	
bw = 0.072 kg (21 d old, Shellenberger 1978)							
fc = 0.0105 $kg \cdot d^{-1}$ (Nagy 1987)							
Mallard *(Anas platyrhynchos)*	Aroclor 1221	LC_{50}	>5000	diet	5 days		Hill et al. 1975
	Aroclor 1232	LC_{50}	>6000				
(10 d old)	Aroclor 1242	LC_{50}	3182				
	Aroclor 1248	LC_{50}	2798				
	Aroclor 1254	LC_{50}	2699				
	Aroclor 1260	LC_{50}	1975				
	Aroclor 1262	LC_{50}	3008				

Appendix XIII (continued). Avian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet ($mg \cdot kg^{-1}$)	Route of Exposure	Exposure Period	Estimated Dose** ($mg \cdot kg^{-1} bw \cdot d^{-1}$)	Reference
Mallard *(Anas platyrhynchos)* bw = 1 kg (Heinz et al. 1989) fc = 0.1 $kg \cdot d^{-1}$ (Heinz et al. 1989)	Aroclor 1254	reproductive success, nest attentiveness: EC (0)	 25	diet	1 month	 2.5	Custer and Heinz 1980
Mallard *(Anas platyrhynchos)* bw = 1.225 kg (study) fc = 0.1 $kg \cdot d^{-1}$ (Heinz et al. 1989)	Aroclor 1242	fertility, embryo mortality, # hatched, chick survival, chick weight gain: EC (0) eggshell thickness: EC (9%)	 150 150	diet	12 weeks	 12.3 12.3	Haseltine and Prouty 1980
Ring-necked pheasant *(Phasianus colchicus)* (14 d old)	Aroclor 1221 Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254 Aroclor 1260 Aroclor 1262	LC_{50} LC_{50} LC_{50} LC_{50} LC_{50} LC_{50} LC_{50}	>5000 3146 2078 1312 1091 1260 1262	diet	5 days		Hill et al. 1975
Red-winged blackbird *(Agelaius phoeniceus)* bw = 0.064 kg (Stickel et al. 1983) fc = 0.0137 $kg \cdot d^{-1}$ (Nagy 1987)	Aroclor 1254	LC_{50}	1500	diet	6 days	321	Stickel et al. 1984
Starling *(Sturnus vulgaris)*	Aroclor 1254	LC_{50}	1500	diet	4 days		Stickel et al. 1984
Brown-headed cowbird *(Molothrus ater)* bw = 0.049 kg (Dunning 1984) fc = 0.011 $kg \cdot d^{-1}$ (Nagy 1987)	Aroclor 1254	LC_{50}	1500	diet	7 days	337	Stickel et al. 1984
Grackle *(Quiscalus quiscula)*	Aroclor 1254	LC_{50}	1500	diet	8 days		Stickel et al. 1984
Bengalese finch *(Lonchura striata)*	Aroclor 1254	LD_{50}		diet	8 weeks	254‡	Prestt et al. 1970
Mourning dove *(Zenaida macroura carolinensis)* bw = 0.16 kg (Terres 1980) fc = 0.0173 $kg \cdot d^{-1}$ (equation, Nagy 1987)	Aroclor 1254	courtship behaviour: EC (96%)	 10	diet	42 days	 1.1	Tori and Peterle 1983
Ring dove *(Streptopelia risoria)* bw = 0.16 kg (Terres 1980) fc = 0.0173 $kg \cdot d^{-1}$ (equation, Nagy 1987)	Aroclor 1254	egg production: EC (17%) eggs hatched: EC (82%) # young fledged: EC (91%)	 10 10 10	diet	9 months	 1.1 1.1 1.1	Peakall et al. 1972
Ring dove *(Streptopelia risoria)* bw = 0.16 kg (Terres 1980) fc = 0.0173 $kg \cdot d^{-1}$ (equation, Nagy 1987)	Aroclor 1254	incubation behaviour: EC (n.r.*)	 10	diet	n.r.*	 1.1	Peakall and Peakall 1973

Appendix XIII (continued). Avian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet (mg•kg^{-1})	Route of Exposure	Exposure Period	Estimated Dose** (mg•kg^{-1}bw· d^{-1})	Reference
Ring dove *(Streptopelia risoria)* FIR = 0.073 kg·kg^{-1}bw· d^{-1} (study)	Aroclor 1254	brain dopamine levels: EC (45%) brain norepinephrine levels: EC (57%) body weight: EC (7%) liver weight: EC (14%)	 10 10 100 100	diet	8 weeks	 0.73 0.73 7.3 7.3	Heinz et al. 1980
White leghorn *(Gallus domesticus)* bw = 1.95 kg (US EPA 1988) fc = 0.132 kg· d^{-1} (US EPA 1988)	Aroclor 1254	egg production: EC (20%)	 5	diet	28 weeks	 0.34	Platonow and Rheinhart 1973
White leghorn *(Gallus domesticus)* bw = 1.95 kg (US EPA 1988) fc = 0.132 kg· d^{-1} (US EPA 1988)	Aroclor 1242	egg production, egg weight, shell thickness, shell weight: EC (0%) egg hatchability: EC (40%)	 80 20	diet	6 weeks	 5.4 1.4	Britton and Huston 1973
White leghorn *(Gallus domesticus)* bw = 1.95 kg (US EPA 1988) fc = 0.132 kg· d^{-1} (US EPA 1988)	Aroclor 1248	egg breaking strength: EC (0%) egg production: EC (13%) egg hatchability: EC (45%)	 20 20 10	diet	8 weeks	 1.4 1.4 0.68	Scott et al. 1975
White leghorn *(Gallus domesticus)* bw = 1.66 kg (Lillie et al. 1975) FC = 0.126 kg· d^{-1} (study)	Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254 Aroclor 1221 Aroclor 1268 Aroclor 5442	egg hatchability: EC (40%) egg hatchability: EC (80%) egg hatchability: EC (87%) progeny mortality: EC (55%) egg hatchability: EC (14%) egg hatchability: EC (0%) egg hatchability: EC (0%) egg hatchability: EC (0%)	 20 20 20 20 20 20 20 20	diet	9 weeks	 1.5 1.5 1.5 1.5 1.5 1.5 1.5 1.5	Lillie et al. 1974

Appendix XIII (continued). Avian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet (mg×kg^{-1})	Route of Exposure	Exposure Period	Estimated Dose** (mg×kg^{-1}bw· d^{-1})	Reference
White leghorn *(Gallus domesticus)*	Aroclor 1232	hatchability: EC (17%)	10	diet	8 weeks	0.76	Lillie et al. 1975
bw = 1.66 kg (study)	Aroclor 1242	hatchability: EC (32%)	10			0.76	
FC = 0.126 kg· d^{-1} (Lillie et al. 1974)		progeny growth: EC (9%)	10			0.76	
	Aroclor 1248	hatchability: EC (36%)	10			0.76	
		progeny growth: EC (8%)	10			0.76	
	Aroclor 1254	hatchability: EC (0%)	20			1.5	
	Aroclor 1016	hatchability: EC (0%)	20			1.5	
White leghorn *(Gallus domesticus)*	Aroclor 1242	embryo mortality: EC (55%)	20	diet	10 weeks	1.4	Ax and Hansen 1975
bw = 1.95 kg (US EPA 1988)	Aroclor 1254	embryo mortality: EC (59%)	20			1.4	
fc = 0.132 kg· d^{-1} (US EPA 1988)	PCB 118	embryo mortality: EC (74%)	20			1.4	
	Purified trichloro-BP	embryo mortality: EC (14%) (n.s.[1])	20			1.4	
	Purified tetrachloro-BP	embryo mortality: EC (17%) (n.s.[1])	20			1.4	
White leghorn *(Gallus domesticus)*	PCB 155	liver weight: EC (82%)	400	diet	3 weeks	42	McKinney et al. 1976
bw = 0.121 kg (14 d old, US EPA 1988)	PCB 136	body weight: EC (18%)	400			42	
fc = 0.0126 kg· d^{-1} (US EPA 1988)		liver weight: EC (33%)	400			42	
	PCB 153	body weight: EC (22%)	400			42	
		liver weight: EC (67%)	400			42	
	PCB 128	body weight: EC (14%)	400			42	
		liver weight: EC (75%)	400			42	
	PCB 169	mortality: EC (100%)	100			10	

Appendix XIII (continued). Avian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet ($mg \cdot kg^{-1}$)	Route of Exposure	Exposure Period	Estimated Dose** ($mg \cdot kg^{-1} bw \cdot d^{-1}$)	Reference
Secondary Consumers:							
Screech owl *(Otus asio)* bw = 0.181 kg (Dunning 1984) fc = 0.025 kg· d^{-1} (Pattee et al. 1988)	Aroclor 1248	# eggs, eggshell thickness, # hatched, # young fledged: EC (0%)	3	diet	8 weeks	0.41	McLane and Hughes 1980
American kestrel *(Falco sparverius)* bw = 0.124 kg (Bloom 1973) fc = 0.05 kg· d^{-1} (Bloom 1973)	Aroclor 1248	shell thickness: EC (5%)	3	diet	6 months	1.2	Lowe and Stendell 1991
American kestrel *(Falco sparverius)*	Arcolor 1254	sperm concentration: EC (23-27%)	33	diet	62-69 days	9-10‡	Bird et al. 1983

* n.r. = not reported

† n.s. = not statistically significant

‡ dose obtained from the study

** estimated dose = $\frac{\text{conc. in food} \cdot \text{fc}}{\text{bw}}$ or (conc. in food · FIR)

fc = food consumption, bw = body weight, FIR = food ingestion rate

Appendix XIV. Mammalian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet (mg×kg^{-1})	Route of Exposure	Exposure Period	Estimated dose** (mg×kg^{-1}bw· d^{-1})	Reference
Primary Consumers:							
White-footed mice *(Peromyscus leucopus)* bw = 0.022 kg (Green and Millar 1987) fc = 0.0034 kg· d^{-1} (Green and Millar 1987)	Aroclor 1254	mortality: EC (33%) reproduction: EC (73%)	 200 200	diet	60 days	 31 31	Merson and Kirkpatrick 1976
White-footed mice *(Peromyscus leucopus)* bw = 0.0019 kg (final bw, study) fc = 0.0034 kg· d^{-1} (Green and Millar 1987)	Aroclor 1254	liver weight: EC (68%)	 25	diet	5 weeks	 4.5	Sanders and Kirkpatrick 1977
Swiss-Webster mice bw = 0.0292 kg (average, study) fc = 0.0054 kg· d^{-1} (equation, US EPA 1988)	Aroclor 1254	liver weight: EC (8%)	 250	diet	8 weeks	 46	Talcott and Koller 1983
White-footed mice *(Peromyscus leucopus)* FIR = 0.142 mg·kg^{-1}bw· d^{-1} (average control, study)	Aroclor 1254	birth interval: EC (35%) # pups born: EC (25%) pup survival: EC (57%)	 10 10 10	diet	9-15 months	 1.4 1.4 1.4	Linzey 1987
White-footed mice *(Peromyscus leucopus)* FIR = 0.142 mg·kg^{-1}bw· d^{-1} (average control, Linzey 1987)	Aroclor 1254	progeny growth: EC (14%) reproduction: EC (n.r.*)	 10 10	diet	2 generations	 1.4 1.4	Linzey 1988
Oldfield mice *(Peromyscus polionotus)* bw = 0.014 kg (Silva and Downing 1995) fc = 0.0019 kg· d^{-1} (Linzey 1987)	Aroclor 1254	First generation - offspring birth weight: EC (13%) weaning weight: EC (13%) Second generation - offspring/month: EC (70%) birth weight: EC (15%) neonate survival: EC (61%)	 5 5 5 5 5	diet	2 generations	 0.68 0.68 0.68 0.68 0.68	McCoy et al. 1995
Big brown bats *(Eptesicus fuscus)*	Aroclor 1260 Note: bats were fed mealworms that had been raised on contaminated wheat bran (10 mg/kg)	reproduction, weight: EC (0)	 6.36	diet	28 days		Clark 1978

Appendix XIV (continued). Mammalian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet ($mg \times kg^{-1}$)	Route of Exposure	Exposure Period	Estimated dose** ($mg \times kg^{-1} bw \cdot d^{-1}$)	Reference
Big brown bats *(Eptesicus fuscus)*	Aroclor 1254 Note: bats were fed mealworms that had been raised on contaminated wheat bran (10 mg/kg)	reduced rate of weight gain: EC (15%)	 9.4	diet	37 days		Clark and Prouty 1977
Cottontail rabbits *(Sylvilagus floridanus)* bw = 1.2 kg (study) fc = 0.237 $kg \cdot d^{-1}$ (study)	Aroclor 1254	hepatic enzyme activity: EC (29%)	 10	diet	10-11 weeks	 2.0	Zepp et al. 1974
Cottontail rabbits *(Sylvilagus floridanus)* bw = 1.2 kg (Zepp et al. 1974) fc = 0.237 $kg \cdot d^{-1}$ (Zepp et al. 1974)	Aroclor 1254	body weight: EC (0%) reproduction: EC (0%)	 10 10	diet	12 weeks	 2.0 2.0	Zepp and Kirkpatrick 1976
Sprague-Dawley rats bw = 0.25 kg (study) fc = 0.0224 $kg \cdot d^{-1}$ (equation, US EPA 1988)	Aroclor 1242	progesterone levels: EC (29%) # pups: EC (100%)	 75 150	diet	36 weeks	 6.7 13.4	Jonsson et al. 1976
Sprague-Dawley rats bw = 0.25 kg (Jonsson et al. 1976) fc = 0.0224 $kg \cdot d^{-1}$ (Jonsson et al. 1976)	Aroclor 1254	reproduction: EC (65%) birth weight: EC (21%)	 300	diet	10 days	 27	Spencer 1982
Fischer rats FIR = 0.095 $kg \cdot kg^{-1} bw \cdot d^{-1}$ (final, study)	Aroclor 1254	liver weight: EC (40%)	 150	diet	10 days	 14	Carter and Mercer 1983
Fischer rats bw = 0.118 kg (final, study) fc = 0.0128 $kg \cdot d^{-1}$ (study)	Aroclor 1254	liver weight: EC (n.r.*) serum cholesterol: EC (n.r.*)	 16 8	diet	4 days	 1.7 0.87	Carter 1984
Fischer rats bw = 0.096 kg (final, study) fc = 0.0125 $kg \cdot d^{-1}$ (study)	Aroclor 1254	liver weight: EC (7%) serum cholesterol: EC (18%)	 8 16	diet	4 days	 1.0 2.1	Carter 1985
Wistar-derived rats bw = 0.312 kg (average control, study) fc = 0.031 $kg \cdot d^{-1}$ (average control, study)	Aroclor 1254	birth weight: EC (11%) pup 14 day body weight: EC (7%) dam liver weight: EC (7%)	 269 2.5 2.5	diet	mating to weaning	 27 0.25 0.25	Overmann et al. 1987[†]

Appendix XIV (continued). Mammalian Toxicity Studies

Organism	PCB Formulation	Effect and Endpoint	Concentration in diet ($mg \times kg^{-1}$)	Route of Exposure	Exposure Period	Estimated dose** ($mg \times kg^{-1} bw \cdot d^{-1}$)	Reference
Sprague-Dawley rats	Aroclor 1242	hormone changes:		diet	5 months		Byrne et al. 1988[†]
		EC (n.r.*)	1			0.09	
bw = 0.238 kg (median, study)	Aroclor 1254	hormone changes:					
fc = 0.021 kg· d^{-1} (median, study)		EC (n.r.*)	1			0.09	
		adrenal weight:					
		EC (20%)	1			0.09	
	Aroclor 1016	hormone changes:					
		EC (n.r.*)	1			0.09	
Sprague-Dawley rats	PCB 28	biochemical changes:		diet	90 days		Chu et al. 1996
		EC (various)	50			4.5	
bw = 0.25 kg (Jonsson et al. 1976)							
fc = 0.0224 kg· d^{-1} (Jonsson et al. 1976)		NOAEL	0.5			0.05	
Secondary Consumers:							
Ferrets *(Mustela putorius furo)*	Aroclor 1242	reproduction:		diet	8 months		Bleavins et al. 1980[‡]
		EC (100%)	20			3.8	
bw = 0.8 kg (study)	Aroclor 1016	reproduction:					
fc = 0.150 kg· d^{-1} (Shaible 1970)		EC (0%)	20			3.8	
Mink *(Mustela vison)*	Aroclor 1242	mortality:					
		LC_{100}	20			3.8	
bw = 0.8 kg (study)		reproduction:					
fc = 0.150 kg· d^{-1} (Shaible 1970)		EC (100%)	5			0.94	
	Aroclor 1016	reproduction:					
		EC (42%)	20			3.8	
Mink *(Mustela vison)*	Aroclor 1254	reproduction:		diet	9 months		Aulerich and Ringer 1977[‡]
		EC (93%)	2			0.27	
bw = 1 kg (US EPA 1993)	Aroclor 1016	reproduction:					
fc = 0.137 kg· d^{-1} (Bleavins and Aurelich 1981)		EC (0%)	2			0.27	
	Aroclor 1221	reproduction:					
		EC (0%)	2			0.27	
	Aroclor 1242	reproduction:					
		EC (0%)	2			0.27	
Mink *(Mustela vison)*	Aroclor 1254	reproduction:		diet	8-14 weeks		Aulerich et al. 1985[‡]
		EC (89%)	2.5			0.37	
bw = 0.93 kg (control, study)	PCB 136	reproduction:					
fc = 0.137 kg· d^{-1} (Bleavins and Aurelich 1981)		EC (0%)	5			0.74	
	PCB 153	reproduction:					
		EC (0%)	5			0.74	
	PCB 169	weight loss:					
		EC (32%)	0.1			0.02	
		mortality:					
		EC (50%)	0.1			0.02	
		reproduction:					
		EC (100%)	0.1			0.02	

Appendix XIV (continued). Mammalian Toxicity Studies

Organism	**PCB Formulation**	**Effect and Endpoint**	**Concentration in diet ($mg \times kg^{-1}$)**	**Route of Exposure**	**Exposure Period**	**Estimated dose** ($mg \times kg^{-1} bw \cdot d^{-1}$)**	Reference
Mink *(Mustela vison)* bw = 1 kg (US EPA 1993) fc = 0.137 $kg \cdot d^{-1}$ (Bleavins and Aurelich 1981)	Aroclor 1254	thyroid activity, adrenal and pituitary gland morphology EC (0%)	 1	diet	8 months	 0.14	Wren et al. 1987‡
Mink *(Mustela vison)* bw = 0.91 kg (initial control, study) fc = 0.137 $kg \cdot d^{-1}$ (Bleavins and Aurelich 1981)	PCB 169	mortality: EC (50%) weight loss: EC (23%) change in organ weight: EC (64%)	 0.05 0.05 0.01	diet	135 days	 0.008 0.008 0.002	Aulerich et al. 1987‡
Mink *(Mustela vison)* FIR = 0.15 $kg \cdot kg^{-1} bw \cdot d^{-1}$ (study)	 Clophen A30 Clophen A50 Clophen A60	reproduction (litter size, kit survival): EC (n.r.*) EC (n.r.*) EC (n.r.*)	 20.25 11.76 0.252	diet	 51 days 88 days 131 days	 3.0 1.8 0.04	cited in Leonards et al. 1995: Den Boer 1984‡ Kihlström et al. 1992‡ Den Boer 1984‡

* n.r. = not reported

** estimated dose = $\frac{\text{conc. in food} \cdot \text{fc}}{\text{bw}}$ or (conc. in food · FIR)

fc = food consumption, bw = body weight, FIR = food ingestion rate

† Consulted study

‡ The minks diet consists primarily of aquatic biota, therefore, mink are not considered appropriate for use in a food chain model leading to the development of soil quality guidelines.

Appendix XV. Consulted Mammalian Carcinogenic and Cancer Promotion Studies

Organism	PCB Formulation (carcinogenic substance)	Effect and Endpoint	Dose ($mg \cdot kg^{-1}$)	Route of Exposure	Exposure Period	Reference
Carcinogenic						
BALB/cj Mice	Aroclor 1254	hepatomas: EC (4%) hepatomas: EC (41%) liver adenofibrosis: EC (100%)	 300 300 300	diet	6 months 11 months	Kimbrough and Linder (1974) cited in WHO 1993
Donryu rats	Kanechlor 400	liver tumours: EC (30%)	 38.5 - 616	diet	400 days	Kimura and Baba (1973) cited in WHO (1993)
Sherman rats	Aroclor 1254	neoplastic nodules: EC (0%) hepatocellular carcinoma: EC (0%) adenofibrosis EC (35%) adenofibrosis EC (95%)	 20, 100, 500 20, 100, 500 100 500	diet	8 months	Kimbrough et al. (1972) cited in WHO (1993)
	Aroclor 1260	neoplastic nodules: EC (0%) hepatocellular carcinoma: EC (0%) adenofibrosis: EC (30%)	 20, 100, 500, 1000 20, 100, 500, 1000 1000			
Sherman rats	Aroclor 1260	hepatocellular carcinomas: EC (14%)	 ~100	diet	21 months	Kimbrough et al. (1975) cited in WHO (1993)
Rats	Aroclor 1242	nodular hyperplasia: EC (0%) hepatomas: EC (16%)	 1, 10, 100 100	diet	12-26 weeks	Calandra (1976) cited in WHO (1993)
	Aroclor 1254	nodular hyperplasia: EC (0%) hepatomas: EC (15%)	 1, 10, 100 100			
	Aroclor 1260	nodular hyperplasia: EC (0%) nodular hyperplasia: EC (10%) hepatomas: EC (28%)	 1, 10 100 100			

Appendix XV (continued). Consulted Mammalian Carcinogenic and Cancer Promotion Studies

Organism	PCB Formulation (carcinogenic substance)	Effect and Endpoint	Dose ($mg \cdot kg^{-1}$)	Route of Exposure	Exposure Period	Reference
Wistar rats	Clophen A30*	hepatocellular carcinoma: EC (n.s.%)	100	diet	800 days	Schaeffer et al. (1984) as cited in WHO (1993)
	Clophen A60*	hepotocellular carcinoma: EC (29%)	100			
Tumour promotion and inhibition:						
Donryu rats (female)	Kanechlor 400 (3'-Me-DAB)	liver neoplasms inhibition EC (n.r.%)	400	diet	26 weeks	Kimura et al. (1976) cited in WHO (1993)
F344 rats (male)	Kanechlor 500 (2-FAA)	Hyperplasia nodules promotion EC (n.r.%)	500, 1000	diet	8 weeks	Tatematsu et al. (1979)
Sprague-Dawley rats (male)	Aroclor 1254 (DENA)	carcinoma neoplasms promotion EC (n.r.%)	100	diet	18 weeks	Preston et al. (1981) as cited in WHO (1993)

n.s. = not significant; n.r.= not reported

*contains no detectable quantities of dibenzofurans

3'-Me-DAB = 3'-methyl-4-dimethylaminoazobenzene; 2-FAA = *N*-2-fluorenylacetamide; DENA = diethylnitrosamine

Appendix XVI. Soil-to-prey Bioaccumulation Factors

Species	PCB Formulation	organ/tissue	PCB concentration in tissue/organ ($mg \cdot kg^{-1}$)	soil concentration ($mg \cdot kg^{-1}$)	Bioconcentration Factor (dw:dw)[a]	Reference
Mammals						
shrew (*Crocidura russula, Sorex araneus*)	∑7PCB[b]	fat in liver	3.3 4.2	0.057 0.27	17.0[c] 4.6[c]	Hendriks et al. (1995)
	∑PCB		9.3 10.3	0.2 1	13.7[c] 3.9[c]	
Vole (*Microtus sp.*)	Arochlor 1260	liver muscle	7.01 4.55	0.67	4.2[c]	Greichus and Dohman (1980)
Field mouse (*Peromuscus sp.*)		liver muscle	11.94 5.52	0.67	5.3[c]	
13-Lined ground squirrel (*Spermophilus sp.*)		liver muscle	8.71 2.89	0.67	3.0[c]	
Lemmings (*Dicrostonyx groenlandicus*)	Aroclor	whole body	0.19	1.6	0.12	INAC 1997
White-footed mouse (*Peromyscus leucopus*)	Aroclor 1254	whole body	2.08 0.96	8060 2480	0.0003 0.0004	Watson et al. 1985

[a] Bioconcentration Factors are for whole body of organism

[b] ∑7PCB = PCB028 + PCB051 + PCB101 + PCB118 + PCB138 + PCB153 + PCB180

[c] Whole body BCF was conservatively estimated by adding the values obtained from the multiplication of each reported organ/tissue concentration by the relative weight of the organ/tissue to whole body (% of body weight, ww basis) (Jongbloed et al. 1996). For BCF calculations mammal livers, muscles and fat were assumed to have average relative weights of 5.0%, 52.9% and 9.4%, respectively (Jongbloed et al. 1996).