



**Rotterdam Convention on the Prior
Informed Consent Procedure for
Certain Hazardous Chemicals and
Pesticides in International Trade**

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Chemical Review Committee
Seventh meeting
Rome, 28 March–1 April 2011
Item 4 (c) (vi) of the provisional agenda*
Technical work: review of notifications
of final regulatory action: pentachlorobenzene

Pentachlorobenzene

Note by the Secretariat

Addendum

Supporting documentation provided by Canada

The Secretariat has the honour to provide, in the annex to the present note, documentation received from Canada to support its notification of final regulatory action for pentachlorobenzene as an industrial chemical. The documentation has not been formally edited.

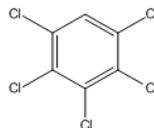
* UNEP/FAO/RC/CRC.7/1.

Annex

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Record 1 of 1

 Molecular Profile Report

 Resources

View Record	Content Type	Source Database	Key
Subscribe	Reference	ChemIndex (CI)	608-93-5
Subscribe	Reference	National Cancer Institute (NCI)	1857
View Record	Reference	NetPro tm Molecular Connections (NETPRO)	MC50882
Subscribe	Safety	Sigma Aldrich MSDS (SAMSDS)	X1003313-5
Subscribe	Sourcing	ChemACX (ACX)	X1003313-5

 Names and Synonyms

Names	Sources
1,2,3,4,5-Pentachlorobenzene	CHEMACX CHEMINDEX NCI NETPRO
Benzene, pentachloro- (8CI9CI)	NCI
Pentachlorobenzene	CHEMACX CHEMINDEX NCI SA-MSDS
OCB	CHEMACX CHEMINDEX NCI
WLN: GR BG CG DG EG	NCI

 Identifiers

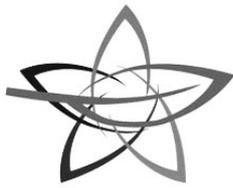
Chemical Identifiers	Type	Sources
X1003313-5	acx_id	SA-MSDS CHEMACX
608-93-5	casrn	CHEMACX CHEMINDEX NETPRO NCI SA-MSDS
MC50882	int_id	NETPRO
608-93-5	mol_rel_id	CHEMACX CHEMINDEX NETPRO NCI SA-MSDS
1857	nsc	NCI

 Properties

Type	Value	Source
Formula	C ₆ HCl ₅	CSCARTRIDGE
MW	250.3371	CSCARTRIDGE
SMILES	C1C=CC(Cl)=C(Cl)C(Cl)=C1Cl	CSCARTRIDGE
Std. InChI	InChI=1S/C6HCl5/c7-2-1-3(8)5(10)6(11)4(2)9/h1H	CSCARTRIDGE
Std. InChIKey	CEOCDNVZRAIOQZ-UHFFFAOYSA-N	CSCARTRIDGE

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ROTTERDAM CONVENTION

SECRETARIAT FOR THE ROTTERDAM CONVENTION
ON THE PRIOR INFORMED CONSENT PROCEDURE
FOR CERTAIN HAZARDOUS CHEMICALS AND PESTICIDES
IN INTERNATIONAL TRADE



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Geneva, 17 December 2010

Subject: PIC procedure – Your submission of notifications of Final Regulatory Action

Dear Mr. Madé,

We are pleased to inform you that the Secretariat has now received two verified notifications from two different PIC regions for:

Pentachlorobenzene

This chemical will now also be sent to the Chemical Review Committee for review and for possible recommendation to the Conference of the Parties for inclusion in Annex III.

To support the work of the Chemical Review Committee at its 7th meeting, we would like to kindly request that you provide the supporting documentation referenced in the notification you have submitted on 28 August 2008, which has been published in the 28th PIC Circular. Documentation referenced in the notifications is outlined below. Please also submit, if available, any other information that was used in the analysis for the decision as those may have been used for section 2.4.2.2. which may not have been included below:

Documentation referenced in the notifications

Pentachlorobenzene

- Prohibition of Certain Toxic Substances Regulations, 2005 (SOR/2005-41) under the Canadian Environmental Protection Act, 1999.
- Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes), 2006 (SOR/2006-279) under the Canadian Environmental Protection Act, 1999.

Mr. Bernard Madé
Director, Chemical Production
Environment Canada
351 St. Joseph Blvd, 17th Floor
Gatineau, Quebec K1A 0H3
Canada

- Follow-up Report on Five PSL 1 Substances for Which There Was Insufficient Information to Conclude Whether the Substances Constitute a Danger to the Environment: 1,2-Dichlorobenzene; 1, 4-Dichlorobenzene; Trichlorobenzenes; Tetrachlorobenzenes; Pentachlorobenzene (2003).
- Risk Management Strategy for Pentachlorobenzene (QCB) and tetrachlorobenzenes (TeCBs). Chemical Control Branch. Environmental Protection Services. June 2005.
- Canadian Environmental Protection Act Priority Substances List Assessment Report: Pentachlorobenzene (1993).
- ChemFinder.com Database and Internet Searching (www.chemfinder.com).
- Groundwater Services, Inc. Chemical properties database (www.gsi-net.com).
- IPCS INCHEM; Concise International Chemical Assessment Document 60; Chlorobenzenes other than hexachlorobenzene: Environmental Aspects (www.inchem.org)

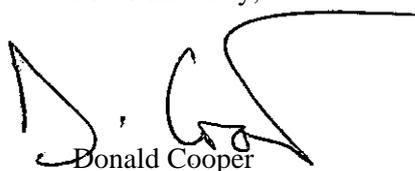
In addition to the above and in line with the process for collecting information on ongoing international trade, we kindly request you to provide, where available, information on whether or not Pentachlorobenzene was manufactured in Canada and, if so, whether it is being exported. You are further requested to provide, if available, the date on which this chemical was last imported into Canada.

The documentation should be provided preferably in English or at least, with a focused summary in English. Please submit this information on or before **12 January 2011**.

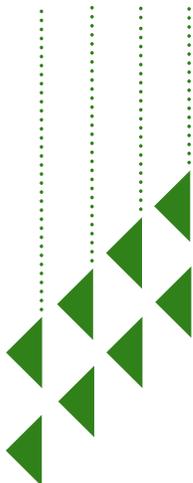
Should you require assistance in compiling this information, or have questions regarding the submission of additional information, please do not hesitate to contact the Secretariat by sending your inquiry to Ms. Neslihan Grasser, Scientific Affairs Officer (e-mail: ngrasser@pic.int).

We would like to take this opportunity to express our gratitude to Canada for your continued support to the work of the Chemical Review Committee.

Yours sincerely,



Donald Cooper
Co-Executive Secretary



Canadian Environmental Protection Act

Priority Substances List
Assessment Report

Pentachlorobenzene



Government
of Canada

Gouvernement
du Canada

Environment
Canada

Environnement
Canada

Health
Canada

Santé
Canada



**Priority Substances List
Assessment Report**

Pentachlorobenzene

Government of Canada
Environment Canada
Health Canada

Aussi disponible en français sous le titre de:
Loi canadienne sur la protection de l'environnement
Liste des substances d'intérêt prioritaire
Rapport d'évaluation: Pentachlorobenzène

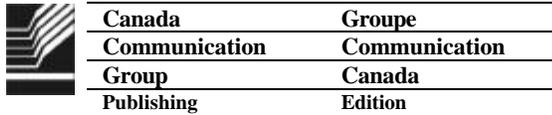
CANADIAN CATALOGUING IN PUBLICATION DATA

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Synopsis

Pentachlorobenzene is not produced in Canada and currently there is no domestic commercial demand for this substance. On the basis of limited data, the 2 most significant sources of entry of pentachlorobenzene into the Canadian environment result from spillage of dielectric fluids and from long-range transport and deposition. Pentachlorobenzene has been detected in samples of air, surface water, rain, sediment and biota collected at various locations in Canada.

Pentachlorobenzene is removed from air and surface water by degradation processes, such as photo-oxidation and biodegradation, but can persist and accumulate under anaerobic conditions in buried sediments and soils.

The highest concentration of pentachlorobenzene detected in Canadian surface waters was over 10 000 times less than the effects threshold estimated for the most sensitive aquatic species identified. For wildlife, the dietary intake of pentachlorobenzene estimated for piscivorous mammals under worst-case conditions was more than 400 times less than the effects threshold estimated on the basis of studies in laboratory mammals. Although significant exposure of benthic organisms to pentachlorobenzene in sediments may be occurring in specific aquatic ecosystems in Canada, adequate data on the toxicological effects on these organisms were not identified. Therefore, it is not possible to determine whether concentrations of this substance in sediments could result in harmful effects to these biota.

Currently, the rates of release of pentachlorobenzene into the environment are low.

Pentachlorobenzene is removed from the atmosphere by photo-oxidation and precipitation and, therefore, is present in low concentrations in air. As such, it is not expected to contribute significantly to global warming or to depletion of stratospheric ozone.

Based on limited available data on concentrations in ambient air, drinking water and food, the average total daily intakes of pentachlorobenzene for various age groups in the general population have been estimated. These average daily intakes are less (by approximately 250 to 1 000 times) than the tolerable daily intake derived on the basis of studies in laboratory animals. For breast-fed infants whose intakes are elevated for only a short period of their life-span, the estimated average total daily intake is 5 times less than the tolerable daily intake. The tolerable daily intake is the intake to which it is believed that a person can be exposed daily over a lifetime without deleterious effect.

Based on these considerations, it has been determined that there is insufficient information to conclude whether pentachlorobenzene is entering the environment in quantities or under conditions that may be harmful to the environment. It has been concluded, however, that pentachlorobenzene is not entering the environment in quantities or under conditions that may constitute a danger to the environment on which human life depends, or to human life or health.

1.0 Introduction

The *Canadian Environmental Protection Act* (CEPA) requires the federal Ministers of the Environment and of Health to prepare and publish a Priority Substances List that identifies substances, including chemicals, groups of chemicals, effluents and wastes that may be harmful to the environment or constitute a danger to human health. The Act also requires both Ministers to assess these substances and determine whether they are "toxic" as defined in section 11 of the Act which states:

“...a substance is toxic if it is entering or may enter the environment in a quantity or concentration or under conditions

- (a) having or that may have an immediate or long-term harmful effect on the environment;
- (b) constituting or that may constitute a danger to the environment on which human life depends; or
- (c) constituting or that may constitute a danger in Canada to human life or health.”

Substances assessed as "toxic" according to section 11 may be placed on Schedule I of the Act, and considered for possible development of regulations, guidelines or codes of practice to control any aspect of their life cycle, from the research and development stage through manufacture, use, storage, transport and ultimate disposal.

The assessment of whether pentachlorobenzene is "toxic", as defined under CEPA, was based on the determination of whether it **enters** or is likely to enter the Canadian environment in a concentration or quantities or under conditions that could lead to **exposure** of humans or other biota at levels that could cause adverse **effects**.

The assessment of whether pentachlorobenzene is "toxic" to human health under CEPA, is based principally on documentation prepared by staff of Health Canada for the International Programme on Chemical Safety (IPCS). Between 1984 and 1987, original data relevant to the assessment of risks to health associated with exposure to the chlorinated benzenes (excluding hexachlorobenzene) were reviewed by staff of Health Canada in the preparation of a draft IPCS Environmental Health Criteria Document (EHC). The current assessment has been updated and expanded to emphasize data most relevant to the assessment of the risks associated with exposure to pentachlorobenzene in the general environment in Canada.

In preparation of the IPCS document, a wide variety of scientific databases were searched to update information provided in earlier contractors' reports including an annotated bibliography on the chlorobenzenes (excluding hexachlorobenzene) by Peter Strahlendorf (1978) and a criteria document on chlorobenzenes (including hexachlorobenzene) by Michael Holliday and Associates (1984a; 1984b). Additional information was identified during peer review of the draft Environmental Health Criteria Document by IPCS focal points and a task group of experts which met in June 1990. More recently, in February 1991, a search was conducted of ENVIROLINE, Chemical Abstracts, Pollution Abstracts, Environmental Bibliography, IRIS, MEDLINE, and BIOSIS databases to identify recent data relevant to assessment in particular of the risks to the population of Canada. Data relevant to assessment of whether pentachlorobenzene is "toxic" to human health obtained after completion of these sections of this report (March 1992) were not considered for inclusion.

Information considered relevant to the assessment of whether pentachlorobenzene is "toxic" to the environment was identified from on-line searches completed in November 1990 of ASFA, BIOSIS, CAB Abstracts, Chemical Abstracts, CESARS, CIS, ENVIROLINE, Hazardous Substances, and IRPTC databases. A summary of information on the environmental toxicity, fate and levels of pentachlorobenzene in the Canadian environment, prepared under contract by Diane Koniacki (November 1991) was also consulted in the preparation of this report. Information received after March 1993 was not included in the environmental sections of this report.

Although review articles were consulted where considered appropriate, original studies that form the basis for the determination of "toxic" under CEPA were critically evaluated by staff of Health Canada (human exposure and effects on human health) and Environment Canada (entry and environmental exposure and effects). The following officials contributed to preparation of the report:

A.M. Bobra (Environment Canada)
D. Boersma (Environment Canada)
L. Brownlee (Environment Canada)
M Giddings (Health Canada)
R. Gomes (Health Canada)
K.M. Lloyd (Environment Canada)
M.E. Meek (Health Canada)

B. Idris and R.G. Liteplo of Health Canada also contributed to the consolidation of the Assessment Report.

In this report, a synopsis that will appear in the *Canada Gazette* is presented. A summary of the technical information that is critical to the assessment, and which is presented in greater detail in unpublished Supporting Documentation, is presented in Section 2.0. The assessment of whether pentachlorobenzene is "toxic", as defined under CEPA, is presented in Section 3.0.

As part of the review and approvals process established by Environment Canada, the environmental sections of this Assessment Report were reviewed by the following: B. Oliver (Zenon Environmental Laboratories); M. Rankin (Dow Chemical Canada Inc.); and H. Rogers (Department of Fisheries and Oceans, Institute of Ocean Sciences). Sections related to the effects on human health were approved by the Standards and Guidelines Rulings Committee of the Bureau of Chemical Hazards of Health Canada. The entire Assessment Report was reviewed and approved by Environment Canada and Health Canada's CEPA Management Committee.

Copies of this Assessment Report and the unpublished Supporting Documentation are available upon request from the following:

Environmental Health Centre
Health Canada
Room 104
Tunney's Pasture
Ottawa, Ontario, Canada
K1A 0L2

Commercial Chemicals Branch
Environment Canada
14th Floor
Place Vincent Massey
351 Saint-Joseph Boulevard
Hull, Quebec, Canada
K1A 0H3

2.0 Summary of Information Critical to Assessment of "Toxic"

2.1 Identity, Properties, Production and Uses

Pentachlorobenzene (Chemical Abstract Service registry number 608-93-5) is a cyclic aromatic hydrocarbon with 5 chlorine atoms substituting for hydrogen atoms in the benzene ring. At room temperature, pentachlorobenzene is a white crystalline solid with a melting point of 86°C (Mackay and Shiu, 1981). Pentachlorobenzene (solid) has an estimated vapour pressure of 0.22 Pa at 25°C (Weast, 1972, 1973). The solubility of pentachlorobenzene in water at 25°C is 0.65 mg/L (Mackay *et al.*, 1992). The log octanol/water partition coefficient (log K_{ow}) for pentachlorobenzene is 5.0 (Mackay *et al.*, 1992). Pentachlorobenzene absorbs infrared radiation, including wavelengths in the 7 to 13 μm region (Sadtler Research Laboratories, 1982).

Pentachlorobenzene can be detected by gas chromatography with either flame ionization or electron capture detection, or by gas chromatography/mass spectrometry (Oliver and Nicol, 1984; Bosma *et al.*, 1988).

Pentachlorobenzene is not produced in Canada (Camford, 1991) and currently there is no domestic commercial demand for this substance. Less than 0.1 kg per year of pure pentachlorobenzene was imported into Canada from the United States for use as a laboratory reagent (Statistics Canada, 1991; Environment Canada, 1992a).

Formerly, pentachlorobenzene as well as trichlorobenzenes and tetrachlorobenzenes were used in combination with polychlorinated biphenyls (PCBs) in dielectric fluids. However, after regulations prohibiting new uses of PCB-containing dielectric fluids were introduced in 1980 (*Canada Gazette*, 1980), the amount of pentachlorobenzene used for this purpose declined considerably. Based on the results of a recent survey (Brien, 1992), small amounts of pentachlorobenzene (40 kg during the first 6 months of 1992) were still imported into Canada in dielectric fluids for use in the maintenance of transformers.

2.2 Entry into the Environment

There are no known natural sources of pentachlorobenzene and currently there is no commercial activity involving this compound in Canada. However, significant quantities of pentachlorobenzene remain in use in dielectric fluids. Based on recent estimates, up to approximately 200 000 kg of pentachlorobenzene are present in dielectric fluids currently in use and up to 1 000 kg of pentachlorobenzene are present in dielectric fluids in storage and destined for disposal by destruction (Environment Canada, 1991a). Since pentachlorobenzene may arise as a by-product or contaminant during the production of other chlorinated organic substances (including tetrachlorobenzenes, hexachlorobenzene, pentachloronitrobenzene, 1,1,1 -trichloroethane, carbon tetrachloride, trichloroethylene, perchloroethylene and ethylene dichloride [US EPA, 1980]), it can enter the environment from releases of these compounds during their storage, use, transport or disposal. Approximately 17 kg per year of pentachlorobenzene may be released into the environment through the use of pentachloronitrobenzene in Canada (Brien, 1992). However, estimates of the amounts of pentachlorobenzene in other chlorinated compounds were not identified.

Pentachlorobenzene has also been identified in the waste streams from the following: pulp and paper mills; iron and steel mills; inorganic and organic chemical plants; petroleum refineries; and activated sludge waste water treatment plants (Rankin, 1993; OME, 1990a, 1990b, 1991a, 1991b, 1991c, 1992a, 1992b; Merriman, 1988; King and Sherbin, 1986; Oliver and Nicol, 1982). The concentrations of pentachlorobenzene ranged from non-detectable (detection limits ranged from 0.01 to 10 ng/L) to 5 400 ng/L; concentrations were highest in effluents from inorganic chemical plants and iron and steel mills. The sources of pentachlorobenzene in these waste streams have not been identified.

No readily available quantitative data on releases of pentachlorobenzene into the Canadian environment were identified. However, amounts released from the most likely sources were estimated and are summarized in Table 1. On the basis of limited data, the 2 most significant sources of entry result from spillage of dielectric fluids and from long-range transport and deposition.

Table 1
Estimated Major Releases of Pentachlorobenzene into the Canadian Environment

Sources	Releases kg/year
Released Due to Dielectric Fluid Spills (after clean-up) ^a	180
Released from Dielectric Fluid Incineration ^b	15
Manufacture of Chlorinated Solvents ^c	
(before April 1992)	(24 to 70)
(after April 1992)	none expected
Use of Chlorinated Solvents ^d	4
From Other Chlorinated Compounds	unknown
Degradation and Metabolism of Other Chlorinated Compounds ^e	unknown
Long-range Transport and Deposition ^f	286
Effluents from Activated Sludge Waste Water Treatment Plants, Pulp and Paper Mills, Iron and Steel Mills, Inorganic and Organic Chemical Plants and Petroleum Refineries ^g	greater than 7
Pesticide Use ⁱ	17
From Landfill Sites ^h	greater than 1

- a. Based on 5.6×10^3 kg for a 10 year period and 67% removal from the initial spill due to recovery efforts (Menzies, 1992); thus, 1.8×10^2 kg/year would be released into the environment. These estimates do not include fugitive releases which can be substantial (Western Research, 1991).
- b. The incineration release assumes a destruction efficiency of 99.99% (Dibbs, 1991); annual quantities are not known. Assumes 100% released in one year.
- c. Before April 1992, approximately 24 to 70 kg were released annually from incineration of waste by-products (based on 1990 production figures [CPI, 1990a, 1990b, 1990c], emission factors by the US EPA (Brooks and Hunt, 1984), incineration efficiency of 99.99% (Environment Canada, 1991b; Jacoff *et al.*, 1986); since April of 1992, chlorinated solvents were no longer produced in Canada other than in one plant which manufactures carbon tetrachloride in Ontario. No pentachlorobenzene is expected to be released into the environment from this source (ICI, 1993).
- d. Based on domestic demand in Canada (Camford, 1991) multiplied by an upper limit concentration of 1 mg/L, and assuming 10% release.
- e. Pentachlorobenzene can be released into the environment, through the metabolism and degradation of other chlorinated compounds, such as lindane (Reed and Forgash, 1970; Karapally *et al.*, 1973; US EPA, 1980) and pentachloronitrobenzene (Kuchar *et al.*, 1969).
- f. Transboundary entry from waste disposal sites (Oliver, 1984, 1985; Oliver and Kaiser, 1986) and in Canadian rainfall (Muir, 1993; Strachan, 1993) and snow (Welch *et al.*, 1991).
- g. Loadings from Ontario iron and steel mills; for the other sectors, loadings are unknown (OME, 1990a, 1990b, 1991a, 1991b, 1991c, 1992a, 1992b). Historically, other sources released pentachlorobenzene into the environment, but due to changing technology, these sources have decreased drastically or been eliminated (Gilbertson, 1979; Alves and Chevalier, 1980).
- h. Loading from only one site (a chemical company landfill in Sarnia, Ontario, used to dispose of chlorinated solvents still bottoms) [King and Sherbin, 1986].
- i. Importation of pentachloronitrobenzene (Brien, 1992), assuming 100% release of pentachlorobenzene as a contaminant.

2.3 Exposure-related Information

2.3.1 Fate

The fate of pentachlorobenzene in the environment is governed by transport processes such as volatilization and adsorption, and transformation processes such as photo-oxidation and aerobic biodegradation. Although pentachlorobenzene is removed from aerobic environments (e.g., air and surface water) by degradation processes, it can persist and accumulate under anaerobic conditions in buried sediments and soils.

Based on its physical and chemical properties, it is anticipated that pentachlorobenzene released into the atmosphere will eventually be distributed between air and water, with smaller amounts in soil and sediment (Mackay *et al.*, 1992). Washout from the atmosphere in rainwater also occurs (Muir, 1993; Strachan, 1993). Pentachlorobenzene can be photo-oxidized in the atmosphere, largely through reactions with hydroxyl (OH) radicals (Atkinson, 1987; Howard *et al.*, 1991). The estimated atmospheric half-life of pentachlorobenzene ranges from 45 to 467 days (1.28 years) [Howard *et al.*, 1991; Mackay *et al.*, 1992; Singleton, 1993]. This half-life is sufficient to permit long-range transport. The presence of pentachlorobenzene in air masses over the Pacific Ocean (Atlas and Schauffler, 1990) suggests that this substance may be transported over long distances.

Based on its physical and chemical properties, pentachlorobenzene released into water would be expected to adsorb to sediment and particulate matter (Kuntz and Warry, 1983; Oliver, 1987b, 1987c; Oliver and Nicol, 1984), with some volatilization to the atmosphere (Oliver, 1984; Oliver and Carey, 1986). The half-life for volatilization of pentachlorobenzene from river water (1 metre deep, flowing at 1 metre/second at 20°C and with a wind velocity of 3 metre/second), calculated according to the method described by Thomas (1982) for compounds of medium to high volatility, was estimated to be 6 hours. In water, biodegradation appears to be the only significant degradation process (Howard *et al.*, 1991). The half-life of pentachlorobenzene in surface water was estimated to range from 194 to 1 250 days (Howard *et al.*, 1991; Mackay *et al.*, 1992). The half-life for the anaerobic biodegradation of pentachlorobenzene in deeper waters ranged from 776 to 1 380 days (Howard *et al.*, 1991).

Pentachlorobenzene is expected to be persistent and immobile in soil. This substance has a *K_{oc}* (organic carbon sorption coefficient) greater than 100 (Kenaga, 1980). Beck and Hansen (1974) calculated a half-life of 270 days for the degradation of pentachlorobenzene in soil, based on the results of a study in which soil was exposed to this substance (10 kg/ha) and stored in plastic covered containers. Half-lives ranging from 194 to 1 250 days have been estimated by Mackay *et al.* (1992).

Based on its physical and chemical properties, pentachlorobenzene is expected to be persistent in sediment (Mackay *et al.*, 1992). Although pentachlorobenzene strongly adsorbs to sediment, some can be removed by resuspension (Oliver, 1984, 1985; Oliver *et al.*, 1989), through biological activities such as bioturbation (Karickhoff and Morris, 1985) or through desorption into pore water and subsequent diffusion into overlying water (Charlton, 1983; Oliver and Charlton, 1984). The half-life for the biodegradation of pentachlorobenzene in sediment was estimated to range from 388 to 1 250 days (Howard *et al.*, 1991; Mackay *et al.*, 1992). On the basis of an analysis of core sediment samples from Lake Ontario and a lake near Kenora, Ontario, pentachlorobenzene has been released into the environment for more than 60 years (Oliver and Nicol, 1982; Oliver, 1984; Durham and Oliver, 1983; Oliver *et al.*, 1989; Muir, 1993) without significant anaerobic degradation in sediment (Oliver and Nicol, 1983).

Reported whole body bioaccumulation factors (BAFs) of pentachlorobenzene vary between 813 (mussel, *Mytilus edulis*) [Renberg *et al.*, 1986] and 20 000 (rainbow trout, *Oncorhynchus mykiss*) [Oliver and Niimi, 1983]. A BAF of pentachlorobenzene of 401 000 was reported for earthworms (*Eisenia andrei*) [Belfroid *et al.*, 1993]. Pentachlorobenzene from sediments can be available to benthic invertebrates. Concentrations of pentachlorobenzene in field populations of oligochaete worms and amphipods were related to levels in the sediments (Fox *et al.*, 1983). The accumulation of pentachlorobenzene by oligochaete worms from sediments in Lake Ontario in laboratory aquaria was demonstrated by Oliver (1987a). The extent of bioaccumulation from sediments appeared to be controlled by concentrations of pentachlorobenzene in the sediment pore water rather than from ingestion of contaminated sediment particles (Oliver, 1987a; Markwell *et al.*, 1989). Although pentachlorobenzene is a bioaccumulative compound, no aquatic food chain magnification has been reported (Thomann, 1989; Oliver and Niimi, 1988).

Air-plant (air-lichen) concentration factors for pentachlorobenzene from the upper Great Lakes region of Ontario ranged from 2.16×10^7 to 3.17×10^7 (Muir *et al.*, 1993). Though there is potential for the uptake of pentachlorobenzene by terrestrial plants through both roots and foliage, it is probably limited (Topp *et al.*, 1986; Trapp *et al.*, 1990).

2.3.2 Concentrations

Pentachlorobenzene has been detected in samples of ambient air, surface water, drinking water, sediment and biota in Canada. Data on the concentrations of pentachlorobenzene in indoor air, estuarine or marine waters, or soil were not identified.

In a survey of ambient air at Windsor and Walpole Island, Ontario, mean concentrations in samples taken between August 1988 and October 1989 were 0.12 ng/m³ in Windsor (detected in 31 of 32 samples) and 0.07 ng/m³ for Walpole Island (detected in 27 of 30 samples), respectively (Environment Canada, 1990). Maximum concentrations at the 2 sites were 0.28 and 0.22 ng/m³ at Windsor and Walpole Island, respectively (detection limit 0.03 ng/m³). The sampling locations in Windsor were located approximately 6 km from a municipal waste incinerator plant in Detroit, Michigan, while Walpole Island is a rural location located 55 km from the same plant. The concentration of pentachlorobenzene in samples of air collected between February and April 1988 at Alert, Northwest Territories, ranged from 0.031 to 0.135 ng/m³ (Patton *et al.*, 1991). Information on concentrations of pentachlorobenzene in indoor air in Canada, or elsewhere, were not identified, except in the vicinity of a limited number of contaminated sites.

The concentration of pentachlorobenzene in samples of rain collected from various locations across Canada between 1987 and 1991 ranged from < 0.01 to 0.09 ng/L; the mean concentration was 0.02 ng/L (Muir, 1993; Strachan, 1993).

The concentration of pentachlorobenzene in samples of water collected from various lakes and rivers in Canada between 1980 and 1991, ranged from non-detectable to 13 ng/L (detection limits ranged from 0.01 to 1 ng/L, depending upon the analytical technique used) [Oliver and Nicol, 1982, 1984; Fox *et al.*, 1983; Oliver, 1984; Chan *et al.*, 1986; Oliver and Kaiser, 1986; Fox and Carey, 1986; Chan and Kohli, 1987; Biberhofer and Stevens, 1987; Carey and Fox, 1987; Oliver and Niimi, 1988; Data Interpretation Group River Monitoring Committee, 1988, 1990; Stevens and Neilson, 1989; Muir, 1993]. All mean concentrations were less than 1.5 ng/L. The maximum concentrations were highest in samples of water obtained in the early 1980s from Lake Ontario at the mouth of the Niagara River (Fox and Carey, 1986; Oliver and Nicol, 1984) and St. Clair River (Oliver and Kaiser, 1986); however, concentrations have subsequently declined. The maximum concentration in samples of water collected from the Niagara River at Niagara-on-the-Lake during 1988 and 1989 was 0.24 ng/L (Data Interpretation Group River Monitoring Committee, 1990).

Information on concentrations of pentachlorobenzene in Canadian drinking water supplies is limited. Concentrations of pentachlorobenzene in the public water supplies of 3 cities in Ontario sampled in 1980, ranged from 0.03 ng/L to 0.05 ng/L with a mean value of 0.04 ng/L (the detection limit was approximately 0.01 ng/L) [Oliver and Nicol, 1982]. In a survey of drinking water from 139 locations in the 4 Atlantic provinces sampled between 1985 and 1988, pentachlorobenzene was not detected in 602 samples analyzed (detection limit = 0.002 µg/L) [Environment Canada, 1989a, 1989b, 1989c, 1989d].

The concentration of pentachlorobenzene in surficial sediments collected from various lakes and rivers in Canada between 1979 and 1991, ranged from non-detectable to 12 000 ng/g (dry weight) [detection limits ranged from 0.1 to 2 ng/g, depending upon the analytical technique] (Environment Canada, 1979; Oliver and Nicol, 1982; Fox *et al.*, 1983; Oliver and Charlton, 1984; Oliver and Bourbonniere, 1985; Oliver and Pugsley, 1986; Charlton and Oliver, 1986; Oliver, 1987b, 1987c; Merriman, 1987; Oliver and Niimi, 1988; Oliver *et al.*, 1989; Kaiser *et al.*, 1990; Welch *et al.*, 1991; OME, 1993). Concentrations were highest in samples obtained in 1985 from the St. Clair River near a waste disposal site at a chemical plant and an effluent outfall from an industrial area of Sarnia. The concentrations ranged from 0.4 to 12 000 ng/g in samples collected in 1985, while in samples obtained in 1990, the levels ranged from 35 to 2 800 ng/g (Oliver and Pugsley, 1986; OME, 1993). The mean concentrations were less than 10 ng/g (dry weight) with the exception of those obtained from Lake Ontario, the St. Clair River, and sites near point sources.

The level of pentachlorobenzene in suspended sediments collected from various locations in southern Ontario and Quebec between 1980 and 1989, ranged from non-detectable to 600 ng/g (dry weight) [detection limits ranged from 0.02 to 5 ng/g, depending upon the analytical technique] (Fox *et al.*, 1983; Kuntz and Warry, 1983; Oliver and Charlton, 1984; Chan *et al.*, 1986; Charlton and Oliver, 1986; Oliver and Kaiser, 1986; Sylvestre, 1987; Merriman, 1987, 1988; Oliver and Niimi, 1988; Data Interpretation Group River Monitoring Committee, 1988, 1990; Oliver *et al.*, 1989; Kaiser *et al.*, 1990). Concentrations of pentachlorobenzene were highest in samples obtained in 1985 from the St. Clair River near a waste disposal site at a chemical plant and an effluent outfall from an industrial area of Sarnia (Oliver and Kaiser, 1986).

The concentration of pentachlorobenzene in samples of biota collected from various lakes and rivers in Canada between 1980 and 1991 ranged from less than 0.01 ng/g to 93 ng/g (wet weight) [Oliver and Nicol, 1982; Oliver and Niimi, 1983, 1988; Fox *et al.*, 1983; Jaffe and Hites, 1986; Oliver, 1987a; Niimi and Oliver, 1989; Metcalfe and Charlton, 1990; Muir *et al.*, 1992; Muir, 1993]. The concentration of pentachlorobenzene in aquatic invertebrates obtained from Lake Ontario near the mouth of the Niagara River ranged from 0.3 to 93 ng/g (wet weight) [Fox *et al.*, 1983; Oliver and Niimi, 1988]. The levels of pentachlorobenzene in macrophytes obtained from the St. Clair River in the industrial area of Sarnia were in the low ng/g range (OME, 1987). Levels in lichens from Ontario were less than 1 ng/g (wet weight) [Muir *et al.*, 1993].

The concentration of pentachlorobenzene in various species of upper trophic level fish obtained in 1980 from Lakes Superior, Huron, Erie, and Ontario ranged from 0.6 to 16 ng/g (wet weight); the highest levels were in fish obtained from Lake Ontario near the Niagara River (Oliver and Nicol, 1982). The concentrations of pentachlorobenzene in various species of lake-dwelling fish collected in northwestern Ontario during 1990 and 1991, were relatively low (< 1 ng/g wet weight) [Muir, 1993].

Between 1980 and 1990, levels of pentachlorobenzene in herring gull eggs (*Larus argentatus*) collected from the Niagara and Detroit Rivers have declined approximately 15-fold, to 1 ng/g (i.e., the limit of detection) [Bishop *et al*, 1992]. Pentachlorobenzene is included by Environment Canada in the routine monitoring of chlorinated organic compounds in wildlife, but is rarely detected at concentrations above 2 ng/g, with the detection limit ranging from 0.1 to 2.0 ng/g (Environment Canada, 1992b). Concentrations above the detection limit are usually observed in samples obtained from known contaminated areas such as the Niagara River. Levels of pentachlorobenzene in samples of narwhal blubber collected between 1982 and 1983 from Pond Inlet, Northwest Territories, ranged from 6.9 to 36.6 ng/g (wet weight) [Muir, 1993].

Information on concentrations of pentachlorobenzene in Canadian food supplies is limited. Pentachlorobenzene was not detected (detection limit = 0.00001 µg/g) in a limited study of fresh food composites from Ontario (Davies, 1988). Composites analyzed included leafy vegetables, fruits, root vegetables (including potatoes), egg/meat and 2% cows' milk which were prepared from samples obtained in 4 retail grocery stores. Pentachlorobenzene was occasionally detected in individual foods (16 samples each of 14 types of foodstuffs) in the U.S. Food and Drug Administration total diet study conducted between April 1982 and April 1986 (Gunderson, 1987). Mean concentrations ranged from not detected (detection limit not stated) to 0.0045 ppm (µg/g). Concentrations were highest in peanut butter and dry roasted peanuts (maximum value, 0.0090 ppm or [µg/g]) with pentachlorobenzene being detected in all 16 samples of each of these foodstuffs.

Pentachlorobenzene has been detected in breast milk. The mean concentration of pentachlorobenzene in the breast milk of Canadian women taken 3 to 4 weeks after parturition was < 1 ng/g (trace) with a maximum value of 1 ng/g. In this survey, the compound was detected in 97% of the 210 samples analyzed (detection limit and sampling period unspecified) [Mes *et al.*, 1986]. In the breast milk of women of the Canadian indigenous population, there were "trace" (< 1 ng/g) amounts of pentachlorobenzene in 17% of the 18 samples analyzed (detection limit not specified) [Davies and Mes, 1987].

2.4 Effects-related Information

2.4.1 Experimental Animals and In Vitro

LD₅₀s for pentachlorobenzene (gavage in peanut oil) are 940 to 1125 mg/kg bw in adult and weanling rats and 1175 and 1 370 mg/kg bw in Swiss Webster mice (Linder *et al.*, 1980). Decreased activity and tremors were observed in both species at sublethal doses; the kidneys, liver and adrenal glands of rats were also enlarged. In some rats, the gastric mucosa was hyperaemic, and a slight reddish fluorescence of the gastrointestinal tract was observed in both rats and mice under ultraviolet light, suggesting porphyria.

Identified information on the toxicity of pentachlorobenzene following short-term administration is restricted to 2 studies in rats (Chu *et al.*, 1983; NTP 1991) and 1 in mice (NTP 1991). In all of these investigations, pentachlorobenzene was administered in the diet. In Sprague-Dawley rats, hepatic enzymes were induced and there were dose-dependent mild histopathological changes in the liver, accumulation of pentachlorobenzene in the fat and liver, and a significant increase in liver weight at the highest dose following administration in the diet of 5, 50 or 500 ppm (mg/kg) [0.59 to 59 mg/kg bw/day] for 28 days. In an NTP study, F344 rats were administered 100, 330, 1 000, 3 300 or 10000 ppm (mg/kg) [9 to 304 mg/kg bw/day for concentrations up to 3 300 ppm). In the highest exposure group, all rats died. There was also depletion of thymic lymphocytes, hyperkeratosis of the forestomach, and forestomach acanthosis in females. At 3 300 ppm (mg/kg) body weights were reduced. There were increases in liver weights at concentrations as low as 100 ppm (mg/kg) in males and centrilobular hepatocellular hypertrophy at higher concentrations (as low as 330 ppm [mg/kg] in males). Increases in relative kidney weights and abnormal hyaline droplet formation in the renal cortical epithelium were observed at all concentrations in male rats (NTP, 1991). In B6C3F₁ mice exposed to the same dietary concentrations (5.2 to 410 mg/kg bw/day), all animals in the 2 highest dose groups (3 300 and 10 000 ppm (mg/kg) died by day 10, having clinical signs of toxicity prior to death. There was a significant increase in liver weights in both sexes at 330 and 1 000 ppm (mg/kg) and mild to moderate depletion of thymic lymphocytes, due to lymphocyte necrosis, in moribund animals or those that died early (NTP, 1991).

Sub-chronic studies in which analyses of body weight gain, survival, clinical signs of toxicity, clinical chemistry, haematology and histopathology of major organs and tissues have been examined in animals exposed to pentachlorobenzene, are restricted to 2 investigations in rats (Linder *et al.*, 1980; NTP 1991) and 1 in mice (NTP, 1991). In all of these bioassays, pentachlorobenzene was administered in the diet. In female

Sherman rats ingesting diets containing 500 ppm (mg/kg) and greater (> 37.5 mg/kg bw/day*) pentachlorobenzene for 100 days, there was an increase in liver weight and hypertrophy of hepatic cells (Linder *et al.*, 1980). There was also an increase in kidney weights and renal hyaline droplet formation in males at 125 ppm (mg/kg) and above (>8.3 mg/kg bw/day*) In addition, at 1 000 ppm (mg/kg) [males 81.1 mg/kg bw/day; females 78.7 mg/kg bw/day*], there was the following: an increase in adrenal weight and focal areas of renal tubular atrophy and interstitial lymphocytic infiltration in males; an increase in kidney weight in females; a decrease in haemoglobin and an increase in white blood cells in both sexes; and decreases in red blood cells and haematocrit in males. The no-observed-effect-level (NOEL) in female rats, derived on the basis of the results of this study, was 250 ppm (mg/kg) [18.2 mg/kg bw/day*]; the lowest-observed-effect-level (LOEL) in males was 125 ppm (mg/kg) [8.3 mg/kg bw/day*].

In a study conducted by the U.S. National Toxicology Program (NTP), F344 rats and B6C3F₁ mice (10 each sex/group) were administered pentachlorobenzene in the diet at concentrations of 0, 33, 100, 330, 1 000 or 2 000 ppm (mg/kg) for 13 weeks (2.2 to 164 mg/kg bw/day in rats; 5.2 to 410 mg/kg bw/day in mice) [NTP 1991]. There were decreases in the mean body weights of male rats at 1 000 ppm [mg/kg] or more, and in females at all concentrations (33 ppm [mg/kg] and above). Absolute and relative liver weights were increased in both sexes at concentrations as low as 33 ppm [mg/kg] in males. At higher concentrations (as low as 330 ppm [mg/kg] for males), there was centrilobular hepatocellular hypertrophy and accumulation of an unidentified yellow-brown pigment in the hepatocytes, possibly containing porphyrins. In males, there were increases in kidney weights and renal histopathological effects at concentrations as low as 100 ppm (mg/kg), while nephrotoxic effects were observed only at higher concentrations in females (increases in kidney weights and histopathological effects at 1 000 ppm [mg/kg] or greater). The spectrum of renal lesions observed in male rats was characteristic of "hyaline droplet nephropathy", while exacerbation of spontaneous nephropathy characterized by renal tubular cell regeneration and homogenous intratubular protein casts was observed in rats of both sexes. The concentration of protein in the urine was increased in male and female rats exposed to dietary concentrations of pentachlorobenzene of 1 000 ppm (mg/kg) and above; this effect was especially prominent in males. Minimal thyroid follicular cell hypertrophy was also observed in male and female rats exposed to 1 000 and 2 000 ppm (mg/kg) pentachlorobenzene. Free thyroxin and total thyroxin concentrations were significantly decreased in exposed male and female rats indicating moderate hypothyroxinemia. At concentrations of 330 ppm (mg/kg) and above in females and 1 000 ppm (mg/kg) and greater in males, there were effects on haematological parameters including decreases

* Average daily intake derived from Figure 1 in Linder *et al.*, 1980.

in haematocrit values, haemoglobin concentrations, erythrocyte count (males), mean corpuscular haemoglobin and mean erythrocyte volume, and mean corpuscular haemoglobin concentration. These observations were consistent with mild to moderate anaemia. The incidence of abnormal sperm in males was also increased at both dietary concentrations at which it was examined (330 and 2 000 ppm [mg/kg]). On the basis of histopathological lesions, the authors considered the NOELs to be 33 ppm (mg/kg) in male rats and 330 ppm (mg/kg) in females (approximately 2.4 and 24 mg/kg bw/day, respectively).

In mice, there were compound-related clinical signs in both sexes including ventral swelling and ruffled fur, at the highest dietary concentration of pentachlorobenzene (2 000 ppm [mg/kg]). Kidney weights were increased at the highest concentrations (330 ppm [mg/kg] and above in males) and functional effects on the thyroid (decreases in total thyroxin concentrations) were observed at all concentrations in both sexes (33 ppm [mg/kg] and above). Liver weights were increased at lower concentrations (as low as 100 ppm [mg/kg] in males). The only exposure-related histological lesion in mice of either sex was centrilobular hepatocellular hypertrophy and minimal necrosis, observed at all concentrations in males and at 330 ppm (mg/kg) [68 mg/kg bw/day] or greater in females. On the basis of the histopathological lesions, the authors considered the NOEL in female mice to be 100 ppm (mg/kg) [approximately 22 mg/kg bw/day]. No NOEL in males could be established (LOEL = 33 ppm [mg/kg] or approximately 5.2 mg/kg bw/day).

Data on the chronic toxicity or carcinogenicity of pentachlorobenzene were not identified.

The results of available studies concerning the embryotoxicity, foetotoxicity and teratogenicity of pentachlorobenzene include one study in rats (Villeneuve and Khera, 1975) and one in mice exposed by gavage in corn oil (Courtney *et al.*, 1977). Although the results of the investigation in rats indicated that pentachlorobenzene is foetotoxic (increased incidence of extra ribs and sternal defects) at doses (50 mg/kg bw/day) below those which induced toxic effects in the mothers (Villeneuve and Khera, 1975), there were no embryotoxic, foetotoxic or teratogenic effects in the offspring of mice at doses which were maternally toxic (50 mg/kg bw/day and above) [Courtney *et al.*, 1977].

In the only identified study on the reproductive toxicity of pentachlorobenzene, Linder *et al.* (1980) reported maternal toxicity (lowest-observed-adverse-effect-level [LOAEL] = 37.5 mg/kg/day*), tremors in suckling pups (LOAEL = 18.2 mg/kg/day), decreases in pre-weaning growth rates and increases in pup mortality in rats at the higher doses.

* Average daily intake derived from Figure 1 in Linder *et al.*, 1980.

Based on limited available data, mutagenicity in *S. typhimurium* with and without metabolic activation, effects on chromosomes in Chinese Hamster ovary cells *in vitro*, and micronuclei in peripheral blood smears in animals from the NTP sub-chronic study, pentachlorobenzene has not been genotoxic (Haworth *et al.*, 1983; NTP, 1991).

2.4.2 Humans

Case reports of adverse effects in individuals, or epidemiological studies of populations exposed to pentachlorobenzene have not been identified.

2.4.3 Ecotoxicology

The acute and chronic toxicity of pentachlorobenzene has been studied in several aquatic species. However, data were not identified concerning the toxicity of this compound to any other biota including sediment- and soil-dwelling organisms, terrestrial invertebrates, aquatic vascular plants, birds or wild mammals. For the water flea (*Daphnia magna*), immobilization was the most sensitive acute endpoint identified, with a 48-hour EC₅₀ of 122 µg/L (Hermens *et al.*, 1984). Based on the results of a 16-day EC₅₀ test, the most sensitive indicator of toxic stress in *Daphnia magna* was a reduction in productivity after exposure to 25 µg/L pentachlorobenzene (Hermens *et al.*, 1984).

The most sensitive endpoint following the acute exposure of fish to pentachlorobenzene, was an 96-hour LC₅₀ of 135 µg/L for the guppy (*Poecilia reticulata*) [van Hoogen and Opperhuizen, 1988]. Larval growth was the most sensitive indicator of toxic stress during early life stage toxicity tests on fish. van Leeuwen *et al* (1990) reported a 28-day no-observed-effect-concentration (NOEC) [for survival, hatching and growth] of 34 µg/L for *Brachydanio rerio*.

Only one study on the toxicity of pentachlorobenzene in plants was identified. Based on the results of a study in which *Lactuca sativa* was grown on soil contaminated with pentachlorobenzene, Hesse *et al* (1991) reported a 14-day EC₅₀ and NOEC (for growth) of 280 and 50 µg/g (dry weight), respectively, when normalized to a soil organic matter content of 10%.

3.0 Assessment of "Toxic" under CEPA

3.1 CEPA 11(a): Environment

Pentachlorobenzene is not produced in Canada and currently there is no domestic commercial demand for this substance. On the basis of limited information, the 2 most significant sources of entry of this substance into the Canadian environment result from spillage of dielectric fluids and from long-range transport and deposition. Pentachlorobenzene has been detected in air, surface water, rain, sediment and biota within Canada. It is removed from air and surface water by degradation processes but can persist and accumulate under anaerobic conditions in buried sediments and soils.

The lowest reported chronic effect level identified for freshwater organisms was 25 µg/L for *Daphnia magna* (16-day EC50 for reduced reproduction). Dividing this value by a factor of 10 to account for interspecies sensitivity, to extrapolate laboratory results to field conditions, and to convert the chronic lowest effect level to a chronic no effect level, results in an estimated effects threshold of 2.5 µg/L. The highest concentration of pentachlorobenzene in surface waters in Canada (i.e., 13 ng/L in a sample obtained from Lake Ontario near the mouth of the Niagara River in 1982), is approximately 200 times less than the calculated effects threshold. Based on more recent data, the concentration of pentachlorobenzene in surface waters in Canada is approximately 10 000 times less than the calculated effects threshold.

Benthic organisms are exposed to pentachlorobenzene in sediments from: the Canadian Great Lakes and their connecting channels; the St. Lawrence River; Hawk and Far Lake, Northwest Territories; and Abercrombie Point, the Economy River, and Truro in Nova Scotia. However, no toxicological data were identified that would permit estimation of an effects threshold for these biota to determine the significance of this exposure. Similarly, because of the lack of information on concentrations and effects in soils (including those at sites where dielectric fluids have been spilled), it is not possible to determine whether soil-dwelling organisms are adversely affected by exposure to the levels of pentachlorobenzene present in Canada.

The potential for adverse effects to wildlife resulting from exposure to pentachlorobenzene has been evaluated based on a worst-case scenario involving the multimedia exposure of mink (*Mustela vison*), a terrestrial mammal having a diet consisting in part of aquatic prey. A daily intake of 126 ng/kg bw/day was estimated for mink living in the area near the mouth of the Niagara River where the levels of pentachlorobenzene in Canadian waters were the highest (Table 2). The uptake of pentachlorobenzene from air and water is negligible compared to that from food. In the absence of toxicological studies on wildlife species, the effects threshold for mink was estimated on the basis of

results of a sub-chronic ingestion study in mice. In this study, the LOEL was considered to be 5.2 mg/kg bw/day (NTP 1991). Dividing this value by a factor of 100 (10 to account for interspecies differences and to extrapolate from the laboratory to the field, and 10 to extrapolate a chronic NOAEL from a sub-chronic LOEL), the effects threshold for wild mammals was estimated to be 52 µg/kg bw/day. Since the estimated intake of pentachlorobenzene by mink is more than 400 times less than the estimated effects threshold, exposure to pentachlorobenzene should have no adverse effect upon mammalian wildlife.

Table 2
Estimated Worst-case Total Daily Exposure of a
Piscivorous Mammal Around the Niagara Region, Ontario

Exposure Route	Environmental Levels ^a	Daily Rate of Consumption (per kg bw) ^b	Daily Intake (ng/kg bw/day)
Air	0.22 ng/m ³	0.55 m ³ /d	0.12
Surface water	0.24 ng/L	0.1 L/d	0.024
Biota (Fish)	0.81 ng/g	155.0 g/d	126
Total	-	-	126

- a. The level in air is the maximum level measured in a rural environment (Walpole Island, Ontario) in 1988-1989 (Environment Canada, 1990); the level in surface water is the maximum level measured in water samples from Niagara-on-the-Lake, 1988-1989 (Data Interpretation Group River Monitoring Committee, 1990); the level in fish is that predicted based on a log BAF of 3.53 for bluegill sunfish and the maximum concentration in surface water of 0.24 ng/L.
- b. Inhalation rate from Stahl (1967); drinking rate from Calder and Braun (1983) and ingestion rate from Nagy (1987), assuming a diet of 75% fish.

On the basis of available data, the concentrations of pentachlorobenzene present in Canadian surface waters are not expected to cause adverse effects in aquatic biota or wildlife. However; toxicological data on the effects of pentachlorobenzene on sediment- and soil-dwelling biota were considered insufficient to determine the significance of concentrations of this compound in these media in Canada. Therefore, the available information is insufficient to conclude whether pentachlorobenzene is entering the environment in quantities or under conditions that may be harmful to the environment.

3.2 CEPA 11(b): Environment on Which Human Health Depends

Although pentachlorobenzene is present as a gas in the troposphere and absorbs infrared radiation in wavelengths ranging from 7 to 13 μm , its low rate of release and relatively rapid removal from the atmosphere by photo-oxidation (half-life ranges from 45 to 467 days) and precipitation, result in low concentrations in the atmosphere ($<0.01 \mu\text{g}/\text{m}^3$). As such, pentachlorobenzene is not expected to contribute significantly to global warming or depletion of stratospheric ozone.

On the basis of available data, it has been concluded that pentachlorobenzene is not entering the environment in quantities or under conditions that may constitute a danger to the environment on which human life depends.

3.3 CEPA 11(c): Human Life or Health

Population Exposure

On the basis of the available data, it is likely that the general population in Canada is exposed to pentachlorobenzene principally in food (Table 3). Owing to the limited data available on concentrations in food in Canada (i.e., a small study of a limited number of food composites in which pentachlorobenzene was not detected [Davies, 1988]), intake from this source was calculated on the basis of the more extensive total diet study conducted in the U.S. (Gunderson, 1987). The total daily intake of pentachlorobenzene for age groups older than 6 months, is estimated to range from $0.0005 \mu\text{g}/\text{kg bw}/\text{day}$ to $0.002 \mu\text{g}/\text{kg bw}/\text{day}$. Based on the concentrations determined in breast milk, total intake for suckling infants aged 0 to 6 months is estimated to be up to several orders of magnitude greater, from $0.01 \mu\text{g}/\text{kg bw}/\text{day}$ to $0.1 \mu\text{g}/\text{kg bw}/\text{day}$.

Table 3
Estimated Daily Intake (µg/kg) of Pentachlorobenzene by the
General Population in Canada

Medium*	Estimated Intake µg/kg bw/day				
	0-6 mois ^a	7 mois - yr ^b	5-11 yr ^c	12-19 yr ^d	20+ yr ^e
Ambient Air ^f	0.00002- 0.00003	0.00002- 0.00004	0.00003- 0.00005	0.00002- 0.00004	0.00002- 0.00004
Drinking Water ^g		0.000002- <0.00005	0.000001- <0.00004	0.0000009- 0.0000009-<0.0001	<0.00007
Breast Milk ^h	0.01-0.1				
Food ⁱ		0.0022	0.0022	0.0011	0.00046
Total Intake	0.01 to 0.1	0.002	0.002	0.001	0.0005

- a. Assumed to weigh 7 kg, breathe 2 m³ of air per day and drink 750 ml of breast milk (as food) per day (Environmental Health Directorate, 1992).
- b. Assumed to weigh 13 kg, breathe 5 m³ of air per day and drink 0.8 litres of water per day (Environmental Health Directorate, 1992).
- c. Assumed to weigh 27 kg, breathe 12 m³ of air per day and drink 0.9 litres of water per day (Environmental Health Directorate, 1992).
- d. Assumed to weigh 57 kg, breathe 21 m³ of air per day and drink 1.3 litres of water per day (Environmental Health Directorate, 1992).
- e. Assumed to weigh 70 kg, breathe 23 m³ of air per day and drink 1.5 litres of water per day (Environmental Health Directorate, 1992).
- f. Based on a range of mean atmospheric concentrations of pentachlorobenzene of 0.00007 µg/m³ to 0.00012 µg/m³ measured at Walpole Island, Ontario, and Windsor, Ontario (Environment Canada, 1990), assuming that concentrations in indoor air are similar to those in ambient air (Environmental Health Directorate, 1992).
- g. Based on a range of mean concentrations of pentachlorobenzene in Canadian drinking water of 0.00004 µg/L (Oliver and Nicol, 1982) to < 0.002 µg/L (Environment Canada, 1989a, 1989b, 1989c, 1989d).
- h. Based on a range of mean concentrations of pentachlorobenzene detected in breast milk (0.1 to 1.0 ng/g) from a Canadian National Survey and in the Canadian indigenous population (Davies and Mes, 1987), and assuming the density of breast milk is equal to 1.0 g/ml.
- i. Estimated daily intake from food based on the levels of pentachlorobenzene in cheese, peanut butter, peanuts, french fries, margarine, vegetable oil, and candy (Gunderson, 1987) and the estimated intake by various age groups of the population of Canada (Environmental Health Directorate, 1992).

* Data were insufficient to estimate intake from soil.

Effects

Available data on the toxicity of pentachlorobenzene are limited. Epidemiological studies of exposed populations are not available and information on chronic toxicity or carcinogenicity in experimental animals has not been identified. Pentachlorobenzene has not been genotoxic in a small number of *in vitro* and *in vivo* studies of a limited range of genetic endpoints.

Pentachlorobenzene has been classified, therefore, in Group V (inadequate data for evaluation) of the classification scheme for carcinogenicity developed for use in the derivation of the *Guidelines for Canadian Drinking Water Quality* (Environmental Health Directorate, 1989).

With the exception of a one-generation study designed to investigate specifically reproductive and developmental effects (Linder *et al.*, 1980), the longest-term studies of the effects of pentachlorobenzene are sub-chronic investigations. In the sub-chronic studies in which pentachlorobenzene was administered in the diet (the principal route of exposure of the general population) to rats (Linder *et al.*, 1980) and to rats and mice (NTP, 1991), the lowest dietary concentration (and associated dose on a body weight basis) at which compound-related effects were observed was that to which male mice were exposed (33 ppm [mg/kg] in the diet; 5.2 mg/kg bw/day) in the NTP bioassay (NTP, 1991). At this concentration, there was minimal to moderate centrilobular hepatocellular hypertrophy and occasional necrosis of hypertrophied hepatocytes (considered to be secondary to the hypertrophy) [NTP, 1991]. On the basis of this LOEL, a tolerable daily intake (TDI) is conservatively (owing to the paucity of available data) derived as follows:

$$\begin{aligned} \text{TDI} &= \frac{5.2 \text{ mg/kg bw/day}}{10\,000} \\ &= 0.0005 \text{ mg/kg bw/day (0.5 } \mu\text{g/kg bw/day)} \end{aligned}$$

where:

- 5.2 mg/kg bw/day is the lowest no-observed-(adverse)-effect level (NO[A]EL) or LO(A)EL in sub-chronic studies conducted to date.
- 10 000 is the uncertainty factor (x 10 for intraspecies variation; x 10 for interspecies variation; x 10 for less than chronic study; x 10 for lack of data on carcinogenicity and chronic toxicity; additional factor of 10 for LOEL rather than NOAEL not incorporated since observed effects at the LOEL were minimal).

In developmental and reproductive studies conducted to date, pentachlorobenzene has not induced adverse effects at concentrations below that upon which the TDI derived above is based (Villeneuve and Khera, 1975; Courtney *et al.*, 1977; Linder *et al.*, 1980).

The total daily intake of pentachlorobenzene for various age groups in the Canadian population is estimated to range from 0.0005 to 0.1 µg/kg bw/day. These estimated average daily intakes are from 5- to 1 000-fold less than the TDI derived above. It should be noted, though, that with the exception of breast-fed infants, whose intakes are elevated for only a short period of their lifespan, the estimated average daily intakes are 250 to 1 000 times less than the TDI.

On the basis of available data, therefore, it has been concluded that pentachlorobenzene is not entering the environment in quantities or under conditions that may constitute a danger in Canada to human life or health.

3.4 Conclusion

Based upon available data, there is insufficient information to conclude whether pentachlorobenzene is entering the environment in quantities or under conditions that may be harmful to the environment. It has been concluded that pentachlorobenzene is not entering the environment in quantities or under conditions that may constitute a danger to the environment on which human life depends, or to human life or health.

4.0 Recommendations

To enable an assessment of the environmental effects of pentachlorobenzene, it is recommended that the following additional data be acquired on a high priority basis:

- (i) concentrations of pentachlorobenzene in soil and sediment, particularly near point sources;
- (ii) toxicity tests (chronic and acute) with benthic organisms representative of those in the Canadian environment, to determine the effects of pentachlorobenzene associated with sediment.

In addition, to permit a more complete assessment of the exposure of the general population in Canada to pentachlorobenzene, additional monitoring data are desirable, particularly for food and breast milk. Investigations of the chronic toxicity and carcinogenicity of pentachlorobenzene in experimental animals are also required to permit a more complete assessment of its toxicity. However, the priority for these studies is considered to be low.

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**Follow-up Report on Five PSL1 Substances for Which There Was
Insufficient Information to Conclude Whether the Substances
Constitute a Danger to the Environment**

**1,2-Dichlorobenzene
1,4-Dichlorobenzene
Trichlorobenzenes
Tetrachlorobenzenes
Pentachlorobenzene**

December 2003

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List of Acronyms and Abbreviations

ACR	acute to chronic ratio
AET	Apparent Effect Threshold
BAF	bioaccumulation factor
BCF	bioconcentration factor
CBz	chlorobenzene
CBzs	chlorobenzenes (1,2-dichlorobenzene, 1,4-dichlorobenzene, trichlorobenzenes, tetrachlorobenzenes, pentachlorobenzene)
CEPA	<i>Canadian Environmental Protection Act</i>
CEPA 1999	<i>Canadian Environmental Protection Act, 1999</i>
CTV	Critical Toxicity Value
CTV _{SED}	Critical Toxicity Value for sediments
DCB	dichlorobenzene
1,2-DCB	1,2-dichlorobenzene
1,4-DCB	1,4-dichlorobenzene
dw	dry weight
EC ₅₀	median effective concentration
EEV	Estimated Exposure Value
ENEV	Estimated No-Effects Value
ENEV _{SED}	Estimated No-Effects Value for sediments
ENEV _{SOIL}	Estimated No-Effects Value for soils
EqP	equilibrium partitioning
HC ₅	hazardous concentration for 5% of exposed organisms
K _{ow}	octanol-water partition coefficient
LC ₅₀	median lethal concentration
LC ₉₀	lethal concentration to 90% of the test organisms
LOEC	Lowest-Observed-Effect Concentration
MDL	method detection limit
NOEC	No-Observed-Effect Concentration
NPRI	National Pollutant Release Inventory
OC	organic carbon
PCBs	polychlorinated biphenyls
PSL	Priority Substances List
PSL1	first Priority Substances List
QCB	pentachlorobenzene
STP	sewage treatment plant
TCB	trichlorobenzene
TeCBs	tetrachlorobenzenes

SYNOPSIS

1,2-Dichlorobenzene (1,2-DCB), 1,4-dichlorobenzene (1,4-DCB), trichlorobenzenes (TCBs), tetrachlorobenzenes (TeCBs) and pentachlorobenzene (QCB), which appeared on the first Priority Substances List (PSL1), were assessed to determine whether these substances should be considered “toxic” as defined under the *Canadian Environmental Protection Act* (CEPA). It was concluded in the PSL1 assessment that these compounds were not “toxic” under Paragraphs 11(b) or 11(c) of CEPA; however, there was insufficient information to conclude whether they could have immediate or long-term harmful effects on the environment, under Paragraph 11(a). Concentration data for these chlorobenzenes (CBzs) in freshwater and marine sediments and soil environments were lacking. Corresponding data reporting effects on benthic and soil-dwelling organisms were also needed to complete this assessment.

Subsequent to the completion of the PSL1 assessments, a revised CEPA, CEPA 1999, came into effect. Paragraph 64(a) of CEPA 1999 has a definition of “toxic” that is similar to that in Paragraph 11(a) under the original CEPA and addresses whether a substance has or may have an immediate or long-term harmful effect on the environment. However, in CEPA 1999, Paragraph 64(a) has been expanded to include effects on biodiversity. Research studies to address data gaps for the CBzs of interest were funded, and emphasis was placed on studies that examined effects on benthic organisms exposed to the CBzs of interest. Additionally, recent literature was reviewed for new data on concentrations in sediment and soil for each of the CBzs under consideration and for information on the effects on organisms resulting from exposure to these compounds.

Both 1,2-DCB and 1,4-DCB are produced in Canada, based on reports from the early 1990s. 1,4-DCB is used more extensively than 1,2-DCB, primarily as an air freshener/deodorizer. During the mid-1990s, 40–45 tonnes of TCBs were expected to be imported into Canada, although imports of TeCBs and QCB were not anticipated.

The primary route of entry for CBzs into Canadian surface waters and associated sediments is via effluents from industrial and sewage treatment plants. 1,2-DCB, 1,4-DCB, TCBs, TeCBs and QCB have been identified in pulp and paper mill effluents. Effluents from iron and steel manufacturing contribute to loadings of TCBs, TeCBs and QCB, while petroleum refinery effluents have been reported to contain TeCBs and QCB. The more highly chlorinated benzenes, particularly hexachlorobenzene, are subject to reductive dechlorination, which may contribute to accumulation of the lower chlorinated homologues (e.g., DCBs and TCBs) in buried sediments. The main source of CBzs to Canadian soils is accidental spillage of industrial chemicals, although CBzs may be added to agricultural soils during amendment with sewage sludge. Industrial emissions to the atmosphere represent another route of entry into the Canadian environment.

Maximum Canadian concentrations of each of the CBzs under consideration in this report were observed in sediment samples collected from the St. Clair River in Ontario. 1,4-DCB was the only CBz detected in Canadian soil samples. The CBzs of interest in this report are known to cause both chronic and acute effects in controlled tests on benthic and soil-dwelling organisms. In general, benthic organisms are more sensitive to the CBzs than soil-dwelling species, based on toxicity studies to date.

Concentrations of the CBzs of interest in the highly contaminated sediments of the St.

Clair River are elevated enough that sensitive benthic organisms could experience adverse effects.

Each of the CBzs under investigation in this report has been estimated to persist in sediment for longer than 2 years. The half-lives of 1,2-DCB, 1,4-DCB, TCBs and TeCBs in soil have been estimated to be approximately 8 months, while QCB's half-life in soil has been estimated to be 2 years. Additionally, TeCBs and QCB are subject to atmospheric transport from its source to remote areas and, therefore, are considered persistent in air. All of the CBzs of interest in this report therefore meet the criteria for persistence as defined in the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000) due to the persistence of these compounds in sediment and soil. The higher chlorinated products, TeCBs and QCB also are persistent in air. The lower chlorinated benzenes (1,2-DCB, 1,4-DCB and TCBs) are not expected to be highly bioaccumulative. However, the TeCBs and QCB do have a high potential to bioaccumulate and meet the bioaccumulation criteria defined in the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000).

There are special concerns about persistent and bioaccumulative substances. Persistent substances can remain in the environment for long periods of time, increasing the probability and the duration of exposure. In addition persistent substances are subject to long-range transport, which results in low-level, widespread contamination. Bioaccumulative substances have the potential to biomagnify, and consequently releases of extremely low concentrations of persistent and bioaccumulative substances may - either alone or in combination with other similar substances - cause severe adverse effects.

Based on the information available, it is concluded that 1,2-DCB, 1,4-DCB and TCBs are not entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity, but TeCBs and QCB are entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity. Therefore, 1,2-DCB, 1,4-DCB and TCBs are not considered "toxic" as defined in Paragraph 64(a) of CEPA 1999. It is, however, concluded that TeCBs and QCB be considered "toxic" as defined in Paragraph 64(a) of CEPA 1999.

QCB and TeCBs are persistent, bioaccumulative, predominantly anthropogenic and are considered "toxic" under Paragraph 64(a) of CEPA 1999, and as such, meet the criteria for Track 1 substances under the Toxic Substances Management Policy. Therefore QCB and TeCB should be subject to virtual elimination of releases to the environment. Since there is currently no commercial demand for QCB and TeCBs in Canada, options to prevent their reintroduction into the Canadian market should be explored.

1.0 INTRODUCTION

1,2-Dichlorobenzene (1,2-DCB), 1,4-dichlorobenzene (1,4-DCB), trichlorobenzenes (TCBs), tetrachlorobenzenes (TeCBs) and pentachlorobenzene (QCB) appeared on the first Priority Substances List (PSL1) of the *Canadian Environmental Protection Act* (CEPA), which was published in the *Canada Gazette*, Part I, on February 11, 1989. Assessments were performed to determine whether these chlorobenzenes (CBzs) should be considered “toxic” as defined under CEPA and were completed in 1993 (Government of Canada, 1993a, 1993b, 1993c, 1993d, 1993e)¹. It was concluded that these substances do not constitute a danger either to the environment on which human life depends or to human life or health, and, therefore, they were not found to be “toxic” under Paragraph 11(b) or 11(c) of CEPA. Additionally, during the period over which the original assessments were conducted, it was determined that concentrations of 1,2-DCB, 1,4-DCB, TCBs, TeCBs and QCB present in Canadian air and surface waters were not likely to cause adverse effects on aquatic biota or wildlife. There was, however, a lack of acceptable data on the effects of these CBzs on benthic and soil-dwelling organisms and on concentrations of CBzs in Canadian soils. Therefore, it was not possible to determine whether environmental harm was occurring due to accumulations of these substances in sediment and soil. The lack of data led to the conclusion that there was insufficient information available on sediments and soils to determine whether these substances should be considered “toxic” under Paragraph 11(a) of CEPA.

A revised CEPA, CEPA 1999, came into effect on March 31, 2000. This new legislation includes Paragraph 64(a), which is similar to Paragraph 11(a) of the original CEPA and addresses whether a substance has or may have an immediate or long-term harmful effect on the environment, although it has been expanded to include effects on biodiversity. CEPA 1999 places more emphasis on pollution prevention, gives consideration to the precautionary principle and requires special treatment of persistent and bioaccumulative substances. Substances that are shown to be both persistent and bioaccumulative, therefore, will be assessed using a more conservative approach than is used for other substances.

As a result of the publication of the PSL Assessment Reports for the CBzs, additional studies were designed and funded. Day *et al.* (1995) and Doe *et al.* (1995) reported data on the toxicity of 1,2-DCB, 1,4-DCB, TCBs and TeCBs to freshwater and marine benthic organisms. Additionally, concentrations of 1,2-DCB, 1,4-DCB, TCBs, TeCBs and QCB were determined in sediments near point sources (i.e., outfalls from sewage treatment plants [STPs] and textile manufacturing plants) in Atlantic Canada. Laboratory studies included toxicity testing to determine effects of exposure to sediments from point source locations (Rutherford *et al.*, 1995). Concentrations of 1,2-DCB and 1,4-DCB were determined in sediment collected by the Ontario Ministry of the Environment near outfalls from chemical manufacturing plants and an STP on the St. Clair River, near Sarnia (DeLuca and Fox, 1995).

A literature search for new data on the CBz substances of interest was performed in 1995 and repeated in 1999. The National Pollutant Release Inventory (NPRI) and Accelerated Reduction/Elimination of Toxics databases supported by Environment Canada were also reviewed for CBz data.

Federal and Ontario government scientists were also requested to provide unpublished data relevant to the assessment of 1,2-DCB, 1,4-DCB, TCBs, TeCBs and QCB under Paragraph 11(a) of CEPA or Paragraph 64(a) of the revised CEPA, CEPA 1999.¹

Recent data confirmed the previous conclusion, which found that environmental harm is not likely to result from the presence of 1,2-DCB, 1,4-DCB, TCBs, TeCBs or QCB in Canadian air and water. The focus of this report, therefore, is the determination of whether accumulations of the specific CBzs in aquatic sediments or soils would harm exposed benthic or soil-dwelling organisms.

2.0 ENTRY CHARACTERIZATION

2.1 Anthropogenic releases in Canada

There are no known natural sources of 1,2-DCB, 1,4-DCB, TCBs, TeCBs or QCB. 1,2-DCB and 1,4-DCB are the only CBzs under examination that are produced in Canada (Table 1). Based on reports from the mid-1990s, 1,4-DCB is used to the greatest extent in Canada (1000 tonnes), primarily as an air freshener/deodorizer (Kovrig, 1996). Survey results projected imports of TCBs to range from 40 to 45 tonnes during the mid-1990s (Camford Information Services, 1991). Imports of TeCBs and QCB into Canada were not anticipated based on survey results (Camford Information Services, 1991).

Both 1,2-DCB and 1,4-DCB releases were reported to the NPRI for 1994–1998. Releases of 1,2-DCB ranged from 0.4 to 0.5 tonnes, while 1,4-DCB releases ranged between 8.1 and 10.4 tonnes (NPRI, 1994, 1995, 1996, 1997, 1998). Releases to air were reported for both compounds. Disposal by incineration was reported to range between 8 and 23 tonnes for 1,2-DCB and between 0.4 and 0.5 tonnes for 1,4-DCB (NPRI, 1994, 1995, 1996, 1997, 1998). Total 1,4-DCB emissions were reported to be 55 tonnes by members of the Canadian Chemical Producers' Association in 1997, down from 116 tonnes in 1993 (Canadian Chemical Producers' Association, 1999). Releases of the other CBzs of interest in this report (TCBs, TeCBs and QCB) have not been reported to the NPRI.

CBzs enter Canadian surface waters and associated sediments primarily via effluents from industrial treatment plants and STPs. The major industrial sectors include chemical manufacturing and textile plants. CBzs have also been observed in effluents from pulp and paper mills (Government of Canada, 1993a, 1993b, 1993c, 1993d, 1993e). Effluents from iron and steel manufacturing contribute to loadings of the TCBs, TeCBs and QCB, while petroleum refinery effluents have been reported to contain TeCBs and QCB (Government of Canada, 1993c, 1993d, 1993e). Reductive dechlorination of the more highly chlorinated benzenes, particularly hexachlorobenzene, may lead to the accumulation of the lower chlorinated homologues (e.g., DCBs and TCBs) in buried sediments (Beurskens *et al.*, 1993a, 1993b).

The main reported source of CBzs to Canadian soils is accidental spillage of industrial chemicals, including dielectric fluids containing polychlorinated biphenyls (PCBs) (Government of Canada, 1993a, 1993b, 1993c, 1993d). Other possible sources include industrial emissions to

¹ The PSL 1 Assessment Report for the chlorobenzenes is available on the following websites: www.hc-sc.ca/hecs-sesc/exsd/psl1.htm or www.ec.gc.ca/substances/ese/eng/psap/PSL1_IIC.cfm.

the atmosphere (Ding *et al.*, 1992) and application of sewage sludge to agricultural soils (Webber and Nichols, 1995).

A few studies have been performed in which CBz concentrations have been reported in sewage sludge. In a study of sewage sludge samples collected from 12 municipalities across Canada, levels of 1,2-DCB and 1,4-DCB were reported (Table 2). Levels of TCBS were below detection limits in sludge from all municipalities. TeCBs and QCB were not included in this survey (Webber and Nichols, 1995). DCB concentrations observed in Canadian sludge samples were lower than concentrations reported in the United States during the 1980s and below levels currently reported in the United Kingdom (Table 2).

3.0 EXPOSURE CHARACTERIZATION

3.1 Environmental fate

3.1.1 Sediment

Under anaerobic laboratory conditions, dechlorination by both biotic and abiotic processes has been observed for all CBzs (Bosma *et al.*, 1988; Peijnenburg *et al.*, 1992; Beurskens *et al.*, 1993b, 1994; Yonezawa *et al.*, 1994). Reported dechlorination half-lives range from only a few days to over 1 year, depending upon the CBz studied and the nature of the sediment used. Mackay *et al.* (1992) estimated average half-lives in surface sediment of approximately 2 years for all of the CBzs considered in this report. CBzs entering the water column generally partition to particulate matter and accumulate in bottom sediments, based on results of fugacity modelling (Mackay *et al.*, 1992) and empirical studies (Oliver and Carey, 1986). CBzs have been shown to persist in sediments for long periods. Oliver and Nicol (1982, 1983) compared the relative proportions of different CBz congeners in surface and subsurface sediments and found little evidence of either microbial oxidation or anaerobic dechlorination of higher chlorinated benzene congeners in Lake Ontario sediments. CBzs have been detected in sediment cores dating back to the early 1900s (Eisenreich *et al.*, 1989; Muir *et al.*, 1995, 1996; Rawn *et al.*, 2000a, 2000b).

Although a large fraction of the di- through pentachlorinated benzenes partition to organic matter, there will be some fraction present in sediment pore waters, either complexed as colloids with dissolved organic matter or as freely dissolved molecules (Di Toro *et al.*, 1991). The uncomplexed molecules may pass through cell membranes, entering organisms during exposure to sediment pore waters. Additionally, direct ingestion of organic carbon (OC) contaminated with CBzs may be an important route of exposure for some benthic organisms.

Desorption studies have suggested that irreversible adsorption occurs in some sediments, and, therefore, equilibrium is not always achieved. Irreversibility in binding of organic compounds is expected to increase with exposure time, although this theory has been questioned by some authors (Kan *et al.*, 1994).

3.1.2 Soil

CBzs may enter surface soil as a result of spills, from sewage sludge additions and via atmospheric deposition from both local and distant industrial sources. Similar to observations in sediments, CBzs partition between particulate and liquid phases. Due to the hydrophobic nature of the di- through penta-CBzs, they are considered to be relatively immobile in soils,

particularly in soils with a high OC content. Volatilization and biodegradation are the main routes of loss for these compounds from soils. Mean half-lives in soil have been estimated to be approximately 8 months for 1,2-DCB, 1,4-DCB, TCBs and TeCBs and 2 years for QCB (Mackay *et al.*, 1992).

Although the bioavailability of these compounds may be reduced in aged soils (Gas Research Institute, 1995), uptake by soil-dwelling organisms can also occur by exposure to freely dissolved forms in pore waters. Another route of exposure for soil-dwelling organisms is ingestion of soil organic matter. In plants, absorption of CBzs may occur via direct uptake by roots or through foliage after volatilization from the soil surface (Trapp *et al.*, 1990; Scheunert *et al.*, 1994; Wang and Jones, 1994).

3.1.3 Biota

Bioconcentration factors (BCFs) and bioaccumulation factors (BAFs) for CBzs have generally been reported on a whole-body basis. BCFs ranging between 270 and 560 were reported for 1,2-DCB in rainbow trout (*Oncorhynchus mykiss*) in laboratory studies (Government of Canada, 1993a). Reported BCFs for 1,4-DCB in rainbow trout ranged between 370 and 1400 (Government of Canada, 1993b). BCFs for TCBs were reported to be between 100 and 4000 in a variety of aquatic biota (Government of Canada, 1993c).

BAFs reported for TeCBs were between 1180 and 135 000 in fathead minnow (*Pimephales promelas*), rainbow trout, guppy (*Poecilia reticulata*) and earthworms (*Eisenia andrei*) (Government of Canada, 1993d). BAFs of 810 and 20 000 were reported for QCB in mussel (*Mytilis edulis*) and rainbow trout, respectively, but a much higher BAF for earthworms (*E. andrei*) (401 000) has also been reported (Government of Canada, 1993e). More recently, Burkhard *et al.* (1997) reported BAFs based on freely dissolved, lipid-normalized concentrations for TCBs, TeCBs and QCB in a number of species. When considered on a whole-body wet weight basis, the BAFs reported by Burkhard *et al.* (1997) were between 427 and 630, between 871 and 1905, and between 6310 and 12 883 for TCBs, TeCBs and QCB, respectively. Bioaccumulation of CBzs generally increases with degree of chlorination.

The estimated log octanol-water partition coefficient ($\log K_{ow}$) for both 1,2-DCB and 1,4-DCB is 3.4. The $\log K_{ow}$ estimates for TCB, TeCBs and QCB are 3.85–4.30, 4.5 and 5.0, respectively (Mackay *et al.*, 1992).

3.2 Environmental concentrations

3.2.1 Sediment near point sources

The CBzs, similar to other non-ionic hydrophobic compounds, partition into the organic matter in sediment. Additionally, the bioavailability of these compounds is inversely proportional to the OC content of the sediment. Therefore, CBz concentrations have been OC normalized in Table 3 using the relationship:

$$\text{OC-normalized concentration (mg/kg dw)} = \frac{C_{\text{CBz}}}{f_{\text{oc}}}$$

where C_{CBz} represents the CBz concentration in whole sediment (mg/kg dw) and f_{oc} represents the OC fraction in the sediment.

The highest reported CBz concentrations in Canadian sediment were observed near industrial sites on the St. Clair River at Sarnia, Ontario, during the 1980s ($\Sigma DCBs$: <MDL–31 $\mu\text{g/g dw}$, or <MDL–2070 $\mu\text{g/g OC}$) (Oliver and Pugsley, 1986). In general, the highest concentrations of the CBzs of interest were measured in samples collected near the Dow Chemical Canada 1st Street Sewer. Sampling stations were located upstream and downstream of chemical manufacturing sites. Fox *et al.* (1983) reported high CBz concentrations (maximum 1,4-DCB concentration = 1.3 $\mu\text{g/g dw}$, or 37 $\mu\text{g/g OC}$) in surficial sediment in Lake Ontario near the Niagara River mouth (Table 3).

More recently, 1,4-DCB was detected in sediment samples collected near municipal wastewater treatment plant effluents (<10–90 ng/g dw, or <0.1–16 $\mu\text{g/g OC}$) in Nova Scotia and New Brunswick (Rutherford *et al.*, 1995). In another recent study, elevated levels of 1,4-DCB (1.7 $\mu\text{g/g dw}$, or 40 $\mu\text{g/g OC}$) were reported in sediment near outfalls from municipal wastewater treatment plants near Victoria, British Columbia (EVS, 1992, 1996). The highest CBz concentrations in Canadian sediment remain in the St. Clair River, adjacent to organic chemical and petrochemical plants near Sarnia (DeLuca and Fox, 1995; Kauss, 1995). Median concentrations of the CBzs of interest in the most contaminated stretch of the river (1–2 km) fell by as much as an order of magnitude between 1984 and 1994 (Oliver and Pugsley, 1986; Bedard and Petro, 1992; Kauss, 1995). However, direct comparisons between the older sampling and more recent work are not possible due to differences in sampling locations and sample collection and analysis techniques.

No recent concentration data for CBzs in sediment from the Niagara River delta were identified. Results of ongoing monitoring of Niagara River water, however, indicate that concentrations of CBzs in river water and suspended sediments have decreased by as much as 10-fold since the early 1980s (Kuntz, 1993), similar to observations in sediments from the St. Clair River. It is, therefore, expected that current CBz concentrations in surface sediment in the Niagara River delta are lower than previously reported for this region (Table 3).

Although 1,4-DCB concentrations were elevated in sediment samples collected near the outfalls of primary STPs at Sarnia, Ontario (DeLuca and Fox, 1995; Kauss, 1995), Halifax, Nova Scotia (Rutherford *et al.*, 1995), and Victoria, British Columbia (EVS, 1992; Chapman *et al.*, 1996), levels were below detection (approximately 0.01 $\mu\text{g/g dw}$, or 2.2 $\mu\text{g/g OC}$) near a secondary STP at Fredericton, New Brunswick, and a lagoon STP at Berwick, Nova Scotia (Rutherford *et al.*, 1995). These results indicate that 1,4-DCB levels are not enriched in sediments near all the Canadian STPs (Rutherford *et al.*, 1995). The CBzs of interest were also below detection limits (approximately 0.01 $\mu\text{g/g dw}$, or 2.4 $\mu\text{g/g OC}$) in sediment collected near the outfalls from textile plants in Caraquet, New Brunswick, Bridgetown, Nova Scotia, and Magog, Quebec (Rutherford *et al.*, 1995).

3.2.2 Long-range transport

Some of the CBzs of interest (TeCBs and QCB) have been reported in lake sediments from both temperate regions and the Canadian Arctic (Eisenreich *et al.*, 1989; Muir *et al.*, 1995, 1996; Allen-Gil *et al.*, 1997; Rawn *et al.*, 2000a, 2000b). Movement of organic compounds to Arctic regions via long-range transport and deposition has been the focus of much study in recent

years. Muir *et al.* (1996) reported that maximum CBz (represented by Σ [QCB + hexachlorobenzene]) concentrations were observed in lake sediments dated to the late 1970s and 1980s, approximately 5–10 years later than maximum concentrations in Lake Ontario. These results are consistent with the cold condensation hypothesis, which explains the movement of organics to remote northern regions (Wania and Mackay, 1993). Allen-Gil *et al.* (1997) reported TeCBs (Σ [1,2,3,4-TeCB + 1,2,4,5-TeCB]) and QCB levels in surface slices of sediment cores collected in Arctic U.S. lakes (mean concentrations: 0.41 ng/g dw and 0.10 ng/g dw, respectively). Total TeCBs (Σ [1,2,3,4-TeCB + 1,2,4,5-TeCB]) concentrations detected in Yukon lake sediments ranged from below the MDL (<0.03 ng/g dw) to 0.54 ng/g dw, and QCB levels ranged from below detection levels (<0.03 ng/g dw) to 1.55 ng/g dw (Rawn *et al.*, 2000b).

3.2.3 Soil

CBzs are expected to partition to solid organic matter (Kenaga and Goring, 1980), although the most bioavailable fraction exists in soil pore waters. The freely dissolved, uncomplexed fraction is inversely proportional to the OC content of the soil (van Gestel and Ma, 1988; Trapp *et al.*, 1990; Hulzebos *et al.*, 1993; CEU, 1995). Therefore, concentrations have been summarized on an OC-normalized basis (Table 4).

CBz concentrations in agricultural soils from 14 sites across Canada were generally below detection limits (approximately 0.05 $\mu\text{g/g dw}$, or 3.5 $\mu\text{g/g OC}$) (Webber, 1994). 1,4-DCB, the only CBz detected in the study, had a detection frequency of approximately 20%, and the maximum reported concentration was 0.14 $\mu\text{g/g dw}$ (4.5 $\mu\text{g/g OC}$).

The CBzs of interest in this report (di- to pentachloro-) have not been measured in sludge-amended soils in Canada. 1,2-DCB and 1,4-DCB concentrations in sludge-amended soils can be estimated using the Ontario Ministry of Environment regulations (OMEE, 1994), in which 40 000 kg/ha was taken to be the maximum addition of sludge to soil (Webber and Nichols, 1995) and 2×10^6 kg/ha was taken to be the mass of soil in the plough layer. Mean concentrations of 1,2- and 1,4-DCB in Canadian sludge were observed to be 0.42 mg/kg dw and 0.87 mg/kg dw, respectively (Webber and Nichols, 1995). This indicates that soil concentrations following sludge treatment would be 0.008 $\mu\text{g/g dw}$ (0.4 $\mu\text{g/g OC}$) and 0.017 $\mu\text{g/g dw}$ (0.9 $\mu\text{g/g OC}$), respectively, assuming an OC content of 2%, based on the following relationship using 1,2-DCB as an example:

$$\begin{aligned} \text{soil concentration following sludge treatment} &= \frac{0.42 \text{ mg/kg dw} \times 40\,000 \text{ kg}}{2 \times 10^6 \text{ kg}} \\ &= 0.008 \text{ mg/kg dw soil (or } \mu\text{g/g dw soil)} \end{aligned}$$

CBz contamination in Canadian soils as a result of atmospheric fallout from nearby industrial activity has not been studied. Elevated concentrations were, however, observed in soils downwind of a highly industrialized area near Niagara Falls, New York (Ding *et al.*, 1992) (Table 4). The Niagara Falls soils have been considered representative of “worst-case” exposure conditions for industrial areas in Canada for the TCBs, TeCBs and QCB.

4.0 EFFECTS CHARACTERIZATION

4.1 Benthic organisms

Effects of 1,2,4-TCB on benthic organisms were reported in two studies during the 1980s. A significant effect on marine macrobenthic community structures was observed in a number of taxa exposed to 1,2,4-TCB at nominal concentrations of 100 and 1000 µg/g dw in sediments (Tagatz *et al.*, 1985). Throughout the study, measured concentrations in sediments ranged between 2.1 and 97 µg/g dw and between 519 and 790 µg/g dw, respectively. Due to the extremely low (<0.02%) OC level in the sediment, conversion to OC-normalized results was not possible for these data (Di Toro *et al.*, 1991).

Clark *et al.* (1987) conducted 10-day bioassays with two marine species using sediment containing approximately 0.3–0.6% OC, spiked with 1,2,4-TCB. No lethality was observed in grass shrimp (*Palaemonetes pugio*) or amphioxus (*Branchiostoma caribaeum*) at nominal concentrations of 10 µg/g dw (approximately 2000 µg/g OC) and 75 µg/g dw (approximately 15 000 µg/g OC).

More recently, Day *et al.* (1995) reported chronic whole-sediment toxicity of 1,2-DCB, 1,4-DCB, 1,2,3-TCB and 1,2,4,5-TeCB to two species of freshwater benthic invertebrates, mayfly (*Hexagenia* spp.) and oligochaete worms (*Tubifex tubifex*), in sediment systems under open static conditions over 21 and 28 days, respectively. Nominal concentrations of 0.5, 5, 50 and 500 µg/g dw were used for 1,2-DCB, 1,4-DCB and 1,2,3-TCB exposures, while 1,2,4,5-TeCB concentrations of 0.5, 5, 50 and 150 µg/g dw were used. Concentrations were measured at the beginning and termination of individual studies (Day *et al.*, 1995). For these studies, a mixture of natural sediment, kaolin and fine silica sand, with an average OC content of 3.93% (3.38–4.45%), was used. Survival and biomass and survival and reproduction were taken to be the endpoints for *Hexagenia* and *T. tubifex* studies, respectively. Survival was not affected by exposure to CBzs in either species, although reduction in growth of *Hexagenia* spp. was observed (18–34%) in the 500 µg/g dw exposures with 1,4-DCB and 1,2,3-TCB and in the 150 µg/g dw exposure with 1,2,4,5-TeCB (Table 5). Reproduction of *T. tubifex* was impaired (64–72%) in the 500 µg/g dw exposure trials with 1,2-DCB, 1,4-DCB and 1,2,3-TCB (Table 5). Because effects on growth and reproduction were observed at only the highest concentration level of each exposure series tested (Table 5) — with the exception of 1,2-DCB and 1,2,4,5-TeCB for *Hexagenia* spp. and *Tubifex tubifex*, respectively, where no effect was observed following exposure to spiked sediments — estimates of traditional endpoint values (e.g., LC₅₀, EC₅₀, LOEC, etc.) were not possible.

Doe *et al.* (1995) reported acute toxicity of 1,2-DCB, 1,4-DCB, 1,2,3-TCB and 1,2,4,5-TeCB based on 10-day exposures of the infaunal amphipod *Rhepoxynius abronius* in marine sediments. The sediment used in this series was a mixture of two natural sediments, resulting in an OC content of 0.55% (Doe *et al.*, 1995). Nominal concentrations of 4, 20, 100 and 500 µg/g dw were used in the 1,2-DCB, 1,4-DCB and 1,2,3-TCB exposure studies. For 1,2,4,5-TeCB, nominal concentrations were 1.2, 6.0, 30 and 150 µg/g dw. Concentrations of each exposure system were measured at the beginning of each study, although measurements at the end of the study were restricted to the highest treatment level only (Doe *et al.*, 1995). Mortality of *R. abronius* was significant at the 100 µg/g dw nominal exposure for 1,2-DCB and 1,2,3-TCB, while significant mortality was reported in the 500 µg 1,4-DCB/g dw and 30 µg 1,2,4,5-TeCB/g

dw nominal exposure systems (Table 6). Doe *et al.* (1995) reported LOEC and NOEC values on a ng/g dw basis for the DCBs, 1,2,3-TCB and 1,2,4,5-TeCB, which have been converted to OC-normalized values in Table 6.

The mode of action of the CBzs is considered non-specific or narcosis (van Wezel *et al.*, 1996a, 1996b). Effect levels for individual CBzs are, therefore, expected to be approximately equal, for a given species, based on molar concentrations (McCarty *et al.*, 1992). Because Day *et al.* (1995) and Doe *et al.* (1995) did not conduct toxicity tests for QCB, estimates of its effect levels were made based on the results of the four CBzs tested, on a molar basis. For example, for 1,2,4,5-TeCB, which has a molecular weight of 215.9 g/mol, the LOEC (8.7 µg/g dw, or 1582 µg/g OC) reported by Doe *et al.* (1995) on a molar basis would be:

$$\begin{aligned} \text{LOEC} &= \frac{1582 \text{ } \mu\text{g/g OC}}{215.9 \text{ g/mol}} \\ &= 7.33 \text{ } \mu\text{g}\cdot\text{mol/g}^2 \end{aligned}$$

The molar LOEC for QCB, therefore, was estimated to be:

$$\begin{aligned} \text{LOEC} &= 7.33 \text{ } \mu\text{g}\cdot\text{mol/g}^2 \times 250.3 \text{ g/mol} \\ &= 1835 \text{ } \mu\text{g/g OC} \end{aligned}$$

based on the TeCB result.

This calculation was repeated for 1,2-DCB, 1,4-DCB and 1,2,3-TCB, and the results for each congener were used to determine a range where effects would be expected due to QCB exposure, for each organism. The range of QCB concentrations over which the lowest effect level would be expected, based on individual congener calculations for *T. tubifex*, was 2750–9010 µg/g OC. The range of lowest QCB concentrations expected to cause effects in *Hexagenia* spp. was estimated to be between 400 and 9510 µg/g OC. LOEC values for *R. abronius* were estimated to range between 1840 and 10 410 µg/g OC. Calculations of LOEC values for *R. abronius* were based solely on initial concentrations.

Additional techniques may be used as part of a weight-of-evidence approach to determine the toxicity of organic compounds to benthic organisms. Di Toro *et al.* (1991) proposed a method for estimating toxicity based on the assumption that equilibrium exists between non-ionic compounds bound to sediment OC and those freely dissolved in pore waters. Concentrations of the freely dissolved compounds in pore waters are considered to be proportional to OC-normalized values in sediment, and, therefore, effects of dissolved compounds on pelagic organisms may be used as a surrogate for effects on benthic organisms exposed to contaminated sediments (Di Toro *et al.*, 1991). Estimates of OC-normalized concentrations in sediment resulting in effects on freshwater and marine organisms were made using the equilibrium partitioning (EqP) method (Table 7). Concentrations were calculated using the relationship:

$$C_{\text{sed}} (\mu\text{g/g OC}) = \frac{C_{\text{diss}} (\mu\text{g/L}) \times K_{\text{ow}} (\text{L/kg})}{1000}$$

where C_{sed} represents the OC-normalized concentration in sediment that is likely to cause an effect in benthic biota, K_{ow} is the octanol-water partition coefficient, C_{diss} represents the corresponding concentration in the freely dissolved state and 1000 is the factor used to convert $\mu\text{g}/\text{kg}$ OC to $\mu\text{g}/\text{g}$ OC.

By using the lowest effect concentrations (e.g., LC_{50} and EC_{50} [16–28 days] data) for pelagic organisms reported in the literature, benthic effects estimates were determined using the EqP method (Table 7). Toxicity data for all the CBZs of interest were available in the literature, although a larger data set exists for freshwater organisms than for marine organisms (Government of Canada, 1993f, 1993g, 1993h, 1993i; Environment Canada, 1994).

4.2 Soil-dwelling biota

CBZs have been shown to affect soil microbial populations. Marinucci and Bartha (1979) reported a 24-hour EC_{50} (respiration) for soil microorganisms at approximately $50 \mu\text{g}$ 1,2,4-TCB/g dw based on nominal concentrations. In another study, microbial respiration was depressed by addition of $1000 \mu\text{g}$ 1,2-DCB/g dw ($67\ 114 \mu\text{g}/\text{g}$ OC) initially, although no effect on respiration was observed by the final day of the 6-day experiment (Walton *et al.*, 1989). Fourteen-day LC_{50} values reported for several earthworm species exposed to 1,4-DCB, 1,2,3-TCB and 1,2,4-TCB ranged from 115 to $563 \mu\text{g}/\text{g}$ dw soil (2592 to $6500 \mu\text{g}/\text{g}$ OC) (Table 8) (Neuhauser *et al.*, 1986; van Gestel and Ma, 1990; van Gestel *et al.*, 1991).

Hulzebos *et al.* (1993) reported 7- to 14-day EC_{50} values (growth) for lettuce (*Lactuca sativa*) exposed to DCBs, TCBs and TeCBs. Measurements using 1,2-DCB were not performed in the study, and no similar data were found in the literature for 1,2-DCB; therefore, the effect concentration for 1,4-DCB was taken to represent that for 1,2-DCB (Table 8). Lettuce was found to be more sensitive to 1,2,3-TCB ($5.8 \mu\text{g}/\text{g}$ dw) and 1,2,4,5-TeCB ($4.2 \mu\text{g}/\text{g}$ dw) than 1,4-DCB, for which the EC_{50} was estimated to be $213 \mu\text{g}/\text{g}$ dw ($19\ 722 \mu\text{g}/\text{g}$ OC). EC_{50} values for individual TeCBs and TCB isomers varied by one and two orders of magnitude, respectively (Table 8).

5.0 ASSESSMENT OF “TOXIC” UNDER CEPA 1999

5.1 CEPA 64(a): Environment

The environmental risk assessment of a PSL substance is based on the procedures outlined in Environment Canada (1997). Analysis of exposure pathways and subsequent identification of sensitive receptors are used to select environmental assessment endpoints (e.g., adverse reproductive effects on sensitive fish species in a community). For each endpoint, a conservative Estimated Exposure Value (EEV) is selected and an Estimated No-Effects Value (ENEV) is determined by dividing a Critical Toxicity Value (CTV) by an application factor. A conservative (or hyperconservative) quotient (EEV/ENEV) is calculated for each of the assessment endpoints in order to determine whether there is potential for ecological harm in Canada. If these quotients are less than one, it can be concluded that the substance poses no significant risk to the environment, and the risk assessment is completed. If, however, the quotient is greater than one for a particular assessment endpoint, then the risk assessment for that endpoint proceeds to an analysis where more realistic assumptions are used and the probability and magnitude of effects are considered. This latter approach involves a more thorough consideration of sources of

variability and uncertainty in the risk analysis.

There are, however, special concerns about persistent and bioaccumulative substances. Persistent substances can remain in the environment for long periods of time, increasing the probability and the duration of potential exposure. Releases of extremely low concentrations of persistent and bioaccumulative substances may lead to accumulations in organisms which can eventually cause adverse effects. Substances that, because of their persistence are subject to long-range transport, are of particular concern because they can result in low-level, widespread contamination. Remote and cold regions, such as the Canadian Arctic, can act as a sink for these compounds. Bioaccumulative substances have the potential to biomagnify through the food chain. Although current science is unable to accurately predict the cumulative effects of exposure to low levels of persistent and bioaccumulative substances on the environment, the potential exists for extensive, irreversible impacts. Assessments of such substances must be performed using a proactive, preventative approach to ensure that widespread cumulative effects do not occur. Therefore, environmental assessments of persistent and bioaccumulative substances require a more conservative approach than that used for other substances, even in situations where a substance is released in a small area and effects appear to be localized.

Conservative methodologies are used for both the exposure and effects characterizations for persistent and bioaccumulative substances. If exposure monitoring data are available, the maximum reported/estimated field concentration is used as the EEV. An additional application factor of 10 was to be used in the effects characterization to calculate the ENEV for persistent and bioaccumulative substances.

5.2 Persistence and bioaccumulation criteria as defined in the Persistence and Bioaccumulation Regulations of CEPA 1999

The persistence criteria as defined under the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000) are available in Appendix 1.

5.2.1 Persistence

5.2.1.1 Sediment

Mackay *et al.* (1992) estimated average half-lives in surface sediment of approximately 2 years for all of the CBzs considered in this report. Additionally, the tetra- and pentachlorinated congeners have been identified in sediments from lakes in both temperate regions and northern Canadian environments (Eisenreich *et al.*, 1989; Muir *et al.*, 1995; Rawn *et al.*, 2000b). The detection of the TeCBs and QCB in northern lake sediments in the absence of nearby sources indicates that these residues are a result of long-range transport, and these CBzs, therefore, meet the criteria for persistence in air. CBzs have been reported in sediments dated to the early 1900s, although maximum concentrations were reported to occur between the 1970s and 1980s. These data are consistent with half-life estimates exceeding 1 year in a variety of sediments.

5.2.1.2 Soil

Mean half-lives in soil have been estimated by Mackay *et al.* (1992) to be approximately 8 months for 1,2-DCB, 1,4-DCB, TCBs and TeCBs and 2 years for QCB. All of the CBzs considered in this report are, therefore, likely to persist in soils under aerobic conditions

(Government of Canada, 1993a, 1993b, 1993c, 1993d, 1993e).

5.2.1.3 Air

TeCBs and QCB have been identified in sediments from lakes in both temperate regions and northern Canadian environments (Eisenreich *et al.*, 1989; Muir *et al.*, 1995; Rawn *et al.*, 2000b). The detection of the TeCBs and QCB in northern lake sediments in the absence of nearby sources indicates that these residues are a result of long-range atmospheric transport, and these CBzs, therefore, meet the criteria for persistence in air.

On the basis of the available information, it can be concluded that all of the CBzs of interest are persistent in soil and sediment according to the criteria stipulated in the Persistence and Bioaccumulation Regulations of CEPA 1999, and TeCBs and QCB are also persistent in air.

5.2.2 Bioaccumulation

BCFs reported on a whole-body basis for 1,2-DCB in rainbow trout (*Oncorhynchus mykiss*) ranged between 270 and 560 in laboratory studies (Government of Canada, 1993a). BCFs for 1,4-DCB in rainbow trout ranged between 370 and 1400 (Government of Canada, 1993b), and BCFs between 100 and 4000 have been reported for TCBs. Reported BAFs for TeCBs range between 1180 and 135 000 (Government of Canada, 1993c, 1993d). BAFs of 810 and 20 000 were reported for QCB in mussel (*Mytilis edulis*) and rainbow trout, respectively, although the BAF determined for earthworms (*Eisenia andrei*) was much higher (401 000) (Government of Canada, 1993e). Both 1,2-DCB and 1,4-DCB have an estimated log K_{ow} of 3.4. The log K_{ow} estimates for TCBs, TeCBs and QCB were 3.85–4.30, 4.5 and 5.0, respectively (Mackay *et al.*, 1992).

On the basis of the available information, it is concluded that the TeCBs and QCB are bioaccumulative substances according to the criteria stipulated in the Persistence and Bioaccumulation Regulations of CEPA 1999.

5.3 Assessment endpoints

The CBzs under investigation in this report (1,2-DCB, 1,4-DCB, TCBs, TeCBs and QCB) were assessed during the PSL1. At that time, it was determined that concentrations of these compounds in Canadian air and surface waters were not likely to cause adverse effects on aquatic biota or wildlife. The assessment endpoints of interest in this report, therefore, are adverse effects on populations of benthic and soil-dwelling species.

5.4 Environmental risk characterization

5.4.1 Sediment

Effects on growth and reproduction were observed in two freshwater species, *Hexagenia* spp. and *Tubifex tubifex*, respectively, following exposure to sediment spiked with 1,2-DCB, 1,4-DCB, 1,2,3-TCB and 1,2,4,5-TeCB (Day *et al.*, 1995). The TCB and TeCB isomers studied were assumed to represent all members of each homologue group. Effects on growth and reproduction were observed at the highest concentration level of each exposure series tested (Table 5), with

the exception of 1,2-DCB and 1,2,4,5-TeCB for *Hexagenia* spp. and *Tubifex tubifex*, respectively. In these studies, no effect was observed following exposure to spiked sediments. Based on these data limitations, estimates of traditional endpoint values (e.g., LC₅₀, EC₅₀, LOEC, etc.) were not possible. For QCB, where no measurements were made, effect concentrations were extrapolated using the measured molar effect concentrations determined for each of the other CBzs. The EqP estimates for benthic organisms were within the range reported for *Hexagenia* spp. and *Tubifex tubifex* by Day *et al.* (1995) for 1,4-DCB and 1,2,4,5-TeCB (Table 5). Measured effects concentrations for 1,2-DCB and 1,2,3-TCB were higher than the EqP estimates. Based on these results, the EqP results were taken to be the conservative CTV_{SED} for 1,2-DCB (1382 µg/g OC), 1,4-DCB (1005 µg/g OC), TCBs (1637 µg/g OC), TeCBs (2846 µg/g OC) and QCB (2500 µg/g OC) for freshwater benthos (Table 9).

Doe *et al.* (1995) reported LOECs for marine organisms exposed to sediment spiked with 1,2-DCB, 1,4-DCB, 1,2,3-TCB and 1,2,4,5-TeCB. The LOEC for QCB was estimated by extrapolating the molar-based values measured for the DCBs, 1,2,3-TCB and 1,2,4,5-TeCB. EqP values were also determined for marine benthic organisms, and comparisons between measured and calculated values were made. The more conservative estimate was taken to be the CTV for marine benthic organisms. The measured LOECs reported by Doe *et al.* (1995) for marine benthos were taken to be the CTV_{SEDS} for 1,2-DCB (1127 µg/g OC) and TeCBs (1582 µg/g OC). The median LOEC for QCB, based on the molar estimates reported by Doe *et al.* (1995), was taken to be the CTV_{SED} (3080 µg/g OC) (Table 9). The EqP value was used as the CTV_{SED} for 1,4-DCB (4999 µg/g OC) and TCBs (504 µg/g OC) (Table 9) in marine systems.

5.4.1.1 Determination of Estimated No-Effects Values (ENEV_{SEDS})

The CTV_{SEDS} determined for each of the CBzs under consideration in this report, in both freshwater and marine sediments, were divided by an application factor, resulting in an ENEV_{SED} to convert chronic lowest-reported-effect levels to no-effect concentrations and to account for extrapolation from laboratory to field conditions and inter- and intraspecies variability (Environment Canada, 1997) (Table 10). An acute to chronic ratio (ACR) of 3:1 (Carlson and Kosian, 1987) was applied to the marine CTV_{SED} for 1,2-DCB, 1,4-DCB, TeCBs and QCB because these CTV_{SEDS} were based on acute studies. Although the freshwater CTV_{SED} for TCB was based on an acute endpoint (LC₉₀) (Lay *et al.*, 1985), the test was performed over a 21-day period using daphnids; therefore, it may be considered a chronic study, and an ACR was thus not deemed necessary. An additional application factor of 10 was used for TeCBs and QCB since these have been shown to be persistent and bioaccumulative compounds according to the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000) and since there are special concerns about the long-term cumulative effects of these types of substances. Freshwater ENEV_{SEDS} ranged from 25 µg/g OC for QCB to 164 µg/g OC for TCBs. Marine ENEV_{SEDS} ranged from 5 µg/g OC for TeCBs to 167 µg/g OC for 1,4-DCB.

5.4.1.2 Determination of risk quotients for sediments

The maximum reported CBz concentrations observed in Canadian sediments were from the St. Clair River (Table 3). These levels were taken to represent the EEVs (EEV_{SEDS}) for freshwater benthic organisms because they are suitable representatives of conservative estimates for freshwater Canadian sediments. Risk quotients were calculated for each CBz of interest in this report using the following relationship, with 1,2-DCB as an example:

$$\begin{aligned}
\text{Quotient} &= \frac{\text{EEV}}{\text{ENEV}} \\
&= \frac{52 \mu\text{g/g OC}}{138 \mu\text{g/g OC}} \\
&= 0.4
\end{aligned}$$

1,2-DCB was found to have a risk quotient less than 1 (Table 11), indicating that 1,2-DCB concentrations are below a level for concern in Canadian freshwater sediments. Risk quotients of greater than 1 were found for 1,4-DCB in approximately 25% of the samples from St. Clair River, while risk quotients of greater than 1 for TCBS were found in 18% of the St. Clair River samples. Risk quotients for TeCBs and QCB exceeded a value of 1 in 28% (11 of 39) and 23% (9 of 39) of samples collected from the St. Clair River, respectively. In general, the highest risk quotients were determined for sediment samples collected within 0.5 km of the Dow Chemical Canada 1st Street Sewer.

Based on the results of the risk analysis for freshwater benthic organisms using conservative EEV data, 1,2-DCB is present in Canadian sediments at concentrations not expected to result in effects on freshwater benthic organisms. 1,4-DCB, TCBS, TeCBs and QCB are present at concentrations in Canadian sediments such that effects on freshwater benthic organisms are possible. The very high concentrations used as freshwater EEV_{SEDS} have been reported at only one site in Canada, the St. Clair River, which indicates that these concentrations do not represent the majority of Canadian sediments.

The lower chlorinated CBzs (e.g., 1,2-DCB, 1,4-DCB and TCBS) have not been reported in sediments from remote areas and do not meet the criteria for both persistence and bioaccumulation in the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000). A less conservative approach to determining EEV_{SEDS} for 1,4-DCB and TCBS in freshwater sediments was, therefore, employed, which takes into account the distribution of concentrations in Canada. Examination of data in Table 3, which summarizes the highest concentrations of 1,4-DCB and TCBS reported in Canadian sediment,² indicates that maximum concentrations in freshwater sediment from other locations in Canada are consistently below ENEVs (i.e., risk quotients <1.0). Furthermore, concentrations of 1,4-DCB and TCBS are less than the ENEVs in approximately 75% of the samples collected from the most highly contaminated 2-km stretch of the St. Clair River. For example, risk quotients calculated using median concentrations for 1,4-DCB and TCBS for this 2-km stretch are 0.4 and 0.5, respectively. This indicates that, in the vast majority of cases, the presence of 1,4-DCB and TCBS in Canadian freshwater sediments will not likely result in adverse effects on freshwater benthic organisms.

The higher chlorinated CBzs, TeCBs and QCB, however, have been reported in freshwater sediments from northern regions of Canada, indicating that they are subject to long-range transport and deposition. TeCBs and QCB also meet the persistence and bioaccumulation requirements of the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000). These factors necessitate the use of a conservative approach in the assessment of TeCBs and QCB.

² Additional data on concentrations in sediment from less contaminated sites elsewhere in Canada are presented in Government of Canada (1993a, 1993b, 1993c, 1993d, 1993e).

Risk quotients for all CBzs of interest in marine systems were less than 1, and, therefore, effects on marine benthic organisms would not be anticipated based on current CBz concentrations in marine sediments.

5.4.2 Soil

Limited effects data for soil-dwelling organisms exist, and no new data were produced during this review of the CBzs. The lowest concentrations causing an effect were, therefore, taken to be the CTV_{SOIL} estimates for each CBz under examination in this report (Table 12).

5.4.2.1 Determination of Estimated No-Effects Values ($ENEV_{SOILS}$)

The CTV_{SOILS} determined for 1,2-DCB, 1,4-DCB, TCBs, TeCBs and QCB in soils were divided by an application factor of 10 to convert chronic lowest-reported-effect levels to no-effect concentrations and to account for extrapolation from laboratory to field conditions and inter- and intraspecies variability (Environment Canada, 1997). An additional factor of 3 was used for all CBzs to account for the limited data on effects on terrestrial organisms. For the TeCBs and QCB, an additional application factor of 10 was used because these compounds are persistent and bioaccumulative. The resulting $ENEV_{SOILS}$ range from 0.6 $\mu\text{g/g OC}$ for 1,2,4,5-TeCB to 157 $\mu\text{g/g OC}$ for 1,2- and 1,4-DCB (Table 12).

5.4.2.2 Determination of risk quotients for soils

The maximum 1,2-DCB and 1,4-DCB concentrations estimated for Canadian sludge-amended soil, based on sludge concentrations reported in Webber and Nichols (1995), were taken to be the EEV for Canadian soil. Individual isomer concentrations reported by Ding *et al.* (1992) (Table 4) were taken to be the EEV_{SOILS} for TCBs and TeCBs. This enabled the calculation of $ENEV_{SOIL}$ values for individual isomers (Table 12). Risk quotients, therefore, were calculated using EEV_{SOIL} and $ENEV_{SOIL}$ determinations for the individual isomers where data were available (Table 13).

The concentrations used to calculate risk quotients for TCBs, TeCBs and QCB are representative of a highly industrialized area in the United States and, therefore, are expected to overestimate the majority of Canadian soil concentrations. These data were selected to represent worst-case conditions that may be observed near highly industrialized areas in Canada. The risk quotients for all CBzs under consideration were below 1, despite these extreme conditions. This indicates that CBz concentrations would not result in effects on Canadian soil-dwelling organisms.

5.5 Sources of uncertainty

During the freshwater sediment exposure studies with 1,2-DCB, 1,4-DCB, TCBs and TeCBs, effects were observed at the highest exposure level only; therefore, the EqP method was used as an additional line of evidence to estimate effect concentrations for freshwater benthic organisms. Effects on benthic organisms as a result of exposure to QCB in sediment, in both freshwater and marine systems, were not reported; therefore, QCB effect levels were estimated by relating effect concentrations of all the other CBzs tested to that of QCB, on a molar basis.

There are a number of sources of uncertainty in this environmental risk assessment. Soil concentration data for the Canadian environment were limited to one study in which highly industrialized areas were not emphasized. Representative data, therefore, were taken from a highly industrialized region in the United States, where such data were available. Additionally, there are limited data in the literature for effects of CBzs on soil-dwelling organisms. An additional application factor was applied to the effects data to account for the small data set in the literature for effects on soil-dwelling organisms.

Although current scientific methods are unable to accurately predict the effects of persistent and bioaccumulative substances on the environment, these substances have been dealt with in a conservative manner in this assessment. Persistent substances can remain in the environment for long periods of time, thereby increasing the probability and duration of exposure relative to compounds that do not persist in the environment. Additionally, substances that are subject to long-range transport are of particular concern, because remote and/or cold regions, such as the Canadian Arctic, can act as a sink for such contaminants. Bioaccumulative substances have the potential to biomagnify through the food chain. Even releases of extremely low concentrations of persistent and bioaccumulative substances can lead to accumulations in organisms having the potential – either alone or in combination with other similar substances - to cause adverse effects on organisms that are continually exposed to them over long periods; therefore, an additional application factor of 10 was applied to the TeCBs and QCB, which are persistent and bioaccumulative substances.

Non-polar halogenated organic compounds occur together in sediments near industrial effluents (Bedard and Petro, 1997). Narcosis is the mode of action of many of these compounds, including CBzs. As a result, a cumulative effect on exposed organisms would be anticipated (McCarty *et al.*, 1992). Toxicity studies using St. Clair River sediments confirmed that exposure to multiple narcotic substances, including QCB, was correlated with increased lethality of mayfly (*Hexagenia limbata*) and midge (*Chironomus tentans*) (Bedard and Petro, 1997). Sediments near the Dow Chemical sewer outfall, where multiple non-polar chlorinated organic compounds were detected, are characterized by low abundance and poor benthic invertebrate diversity (Bedard and Petro, 1997).

Losses of the volatile compounds, such as TCBs, were observed in toxicity studies relative to field samples (Bedard and Petro, 1997), indicating that many toxicity studies may result in an underestimation of toxicity.

ENEV_{SEDS} determined in this assessment were compared with hazard-based assessments and effect threshold studies. Quantitative structure–activity relationships and EqP models were used by van Leeuwen *et al.* (1992) to estimate concentrations of CBzs that would not be expected to affect 95% of the species in benthic communities (marine or freshwater). These concentrations, known as HC₅s because they are expected to be hazardous concentrations for 5% of the exposed species, were estimated to be approximately 107 µg/g OC for both 1,2- and 1,4-DCB and 115 µg/g OC for TCBs. These values are within an order of magnitude of the freshwater and marine ENEV_{SEDS} developed in this analysis. The HC₅ for TCBs (115 µg/g OC) compares extremely well with the ENEV_{SEDS} developed for TCBs in the present assessment (164 µg/g OC [freshwater] and 50 µg/g OC [marine]) (Table 10). The HC₅s estimated for TeCBs and QCB (119 µg/g OC and 120 µg/g OC, respectively), however, were 1–2 orders of magnitude higher than the freshwater and marine ENEV_{SEDS} determined in this analysis (Table 10). The lower ENEV_{SED} values for the TeCBs and QCB determined in this assessment are a result of the

conservative approach taken with persistent and bioaccumulative substances. Using another approach, marine Apparent Effect Thresholds (AETs) were developed in the Puget Sound area of Washington State, using data from paired sediment chemistry–effects measurements performed with field sediment samples (Barrick *et al.*, 1988). AET estimates for 1,2-DCB, 1,4-DCB and TCBS (2 µg/g OC, 16 µg/g OC and 3 µg/g OC, respectively) were lower by an order of magnitude or more than marine ENEV_{SEDS} developed in this assessment. The AET method has, however, been criticized because the results may be strongly influenced by the presence of unmeasured contaminants (Chapman, 1989).

5.6 Conclusions

Concentrations of 1,2-DCB, 1,4-DCB, TCBS, TeCBs and QCB in Canadian soil are unlikely to be causing harm to populations of soil-dwelling organisms. However, it is possible that concentrations of 1,4-DCB, TCBS, TeCBs and QCB in sediment from the St. Clair River near Sarnia may be harming benthic organisms.

Of the CBzs under consideration, only TeCBs and QCB meet the criteria for both persistence and bioaccumulation specified in the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada, 2000). There are special concerns about persistent and bioaccumulative substances. Persistent substances can remain in the environment for long periods of time, increasing the probability and the duration of exposure. In addition persistent substances are subject to long-range transport, which results in low-level, widespread contamination. Bioaccumulative substances have the potential to biomagnify, and consequently releases of extremely low concentrations of persistent and bioaccumulative substances may - either alone or in combination with other similar substances - cause severe adverse effects.

CEPA 64(a): Based on available data, it is concluded that 1,2-dichlorobenzene, 1,4-dichlorobenzene and the trichlorobenzenes are not entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity. Therefore 1,2-dichlorobenzene, 1,4-dichlorobenzene and the trichlorobenzenes are not considered “toxic” as defined under Paragraph 64(a) of CEPA 1999.

Based on available data, tetrachlorobenzenes and pentachlorobenzene are entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity and that the tetrachlorobenzenes and pentachlorobenzene are considered “toxic,” as defined under Paragraph 64(a) of CEPA 1999.

6.0 CONSIDERATIONS FOR FOLLOW-UP

It is recommended that both TeCBs and QCB be added to the List of Toxic Substances (Schedule I) of CEPA 1999.

QCB and TeCBs are persistent, bioaccumulative, predominantly anthropogenic and are considered “toxic” under Paragraph 64(a) of CEPA 1999, and as such, meet the criteria for Track 1 substances under the Toxic Substances Management Policy. Therefore QCB and TeCB should be subject to virtual elimination of releases to the environment. Since there is currently no commercial demand for QCB and TeCBs in Canada, options to prevent their reintroduction into the Canadian market should be explored.

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Table 1. Summary of information on production and uses of CBzs in Canada¹

CBz	Produced in Canada	Primary applications
1,2-DCB	yes	Industrial cleaning solvents
1,4-DCB	yes	Air fresheners/deodorizers Moth and bird repellents
TCBs	no	Solvents in textile industry Chemical manufacturing Transformer maintenance
TeCBs	no	Transformer maintenance
QCB	no	Laboratory reagent

¹ Based on data reported by Camford Information Services (1991); data applicable to 1995.

Table 2. Concentration of CBzs in sewage sludge

Country	Reference	Median concentration ($\mu\text{g}/\text{kg dw}$)				
		1,2-DCB	1,4-DCB	ΣTCBs	ΣTeCBs	QCB
Canada	Webber and Nichols (1995)	<MDL ¹ – 451 ²	1–810 ²	ND ³	NA ⁴	NA ⁴
United States	Jacobs <i>et al.</i> (1987)	6455	20205	4045	ND ³	Not reported
United Kingdom	Rogers <i>et al.</i> (1989)	7900	9800	780	80	<MDL ¹
United Kingdom	Wang and Jones (1994)	2310	1120	558	67	47

¹ Below method detection limit.

² Range of median values was determined for each of 12 sludge treatment plants.

³ Not detected.

⁴ Not analysed.

⁵ Data reported as means rather than medians.

Table 3. Recent OC-normalized CBz concentrations, reported as median values, in Canadian sediments near point sources

	Median concentration (range) (µg/g)					Year collected	Reference
	1,2-DCB	1,4-DCB	ΣTCBs	ΣTeCBs	QCB		
Industrial, particularly chemical manufacturing							
St. Clair River near Sarnia, Ontario ¹	1.5 (0.2–52)	37 (2–522)	25 (1–539)	3.5 (0.1–320)	12 (0.3–601)	1994	DeLuca and Fox (1995); Kauss (1995)
Niagara River delta, Lake Ontario ²	0.64 (0.51–1.5)	5.2 (<2.2–37)	3.4 (2.5–9.8)	2.9 (2.0–9.1)	1.1 (0.7–3.4)	1981	Fox <i>et al.</i> (1983)
Textile mills							
Atlantic Canada (3 sites) ¹	<2.4 (<2.4)	<2.4 (<2.4)	<2.4 (<2.4)	<2.4 (<2.4)	<2.4 (<2.4)	1994	Rutherford <i>et al.</i> (1995)
STPs							
Victoria, B.C. ¹	0.06 (0.01–0.30)	1.0 (0.1–40)	0.02 (0.01–0.3)	–	–	1991	EVS (1992)
Sarnia, Ontario ¹	0.34 (0.13–0.83)	2.0 (0.40–7.5)	2.0 (0.20–4.9)	0.09 (0.07–0.42)	0.09 (0.07–0.42)	1994	Kauss (1995)
Halifax, Nova Scotia ¹	<2.2 (<2.2)	0.4 (<0.1–16)	<2.2 (<2.2)	<2.2 (<2.2)	<2.2 (<2.2)	1994	Rutherford <i>et al.</i> (1995)
Other Atlantic sites, 1994 ³	<2.2 (<2.2)	<2.2 (<2.2)	<2.2 (<2.2)	<2.2 (<2.2)	<2.2 (<2.2)	1994	Rutherford <i>et al.</i> (1995)

¹ OC content ranged between 0.2% and 10.1%.

² OC content assumed to be 3.5%, based on analyses of delta samples (Mudroch, 1983).

³ Fredericton, New Brunswick, and Berwick, Nova Scotia.

Table 4. Median OC-normalized CBz concentrations in soils

	Median concentration (range) ($\mu\text{g/g}$)								Year collected	Reference	
	1,2-DCB	1,4-DCB	1,3,5-TCB	1,2,4-TCB	1,2,3-TCB	1,2,3,5-/1,2,4,5-TeCB	1,2,3,4-TeCB	QCB			
Near agricultural source (Canada) ¹	<3.5 (<3.5)	<3.5 (<3.5–4.5)	Not measured	<3.5 (<3.5)	Not measured	Not measured	Not measured	Not measured	Not measured	early 1990s	Webber (1994)
Near industrial source (Niagara Falls, NY) ²	Not measured	Not measured	Not measured	0.127 (0.060–0.255)	0.027 (0.012–0.050)	0.231 (0.120–0.400)	0.156 (0.065–0.325)	0.052 (0.024–0.085)		late 1980s	Ding <i>et al.</i> (1992)

¹ OC-normalized (OC content ranged between 0.1% and 3.8%).

² Normalized assuming 2% OC content.

Table 5. Percentage of freshwater test populations (*Hexagenia* spp. and *Tubifex tubifex*) affected by CBz exposure ($\mu\text{g/g}$ OC normalized¹) after 21-day and 28-day exposures, respectively (Day *et al.*, 1995)

CBz	<i>Hexagenia</i> spp.				<i>Tubifex tubifex</i>			
	Reduction in growth observed (% of affected organisms) ²	Concentration			Reduction in number of young produced (% of affected organisms) ²	Concentration		
		Nominal ($\mu\text{g/g}$ dw)	Initial ($\mu\text{g/g}$ OC)	Final ($\mu\text{g/g}$ OC)		Nominal ($\mu\text{g/g}$ dw)	Initial ($\mu\text{g/g}$ OC)	Final ($\mu\text{g/g}$ OC)
1,2-DCB ³	none	500	3789	218	67	500	4448	1871
1,4-DCB	25	500	3677	234	64	500	3187	1573
1,2,3-TCB	34	500	6845	1444	72	500	6556	5947
1,2,4,5-TeCB ⁴	18	150	3128	2012	none	150	3981	3266

¹ % OC = $3.93\% \pm 0.56\%$.

² Statistical significance ($P < 0.05$) of effects, relative to solvent controls, demonstrated using Dunnett's test.

³ No effect on growth of *Hexagenia* spp. was observed at the highest exposure concentration of 1,2-DCB, initial = 3789 $\mu\text{g/g}$ OC, final = 218 $\mu\text{g/g}$ OC.

⁴ No effect on growth of *T. tubifex* was observed at the highest exposure concentration of 1,2,4,5-TeCB, initial = 3981 $\mu\text{g/g}$ OC, final = 3266 $\mu\text{g/g}$ OC.

Table 6. Lowest concentrations of CBzs causing effects on the marine amphipod *Rhepoxynius abronius* (Doe *et al.*, 1995)

CBz	Observed % mortality ¹	Concentration			NOEC ³ (µg/g OC ²)	LOEC ³ (µg/g OC ²)
		Nominal (µg/g dw)	Initial (µg/g OC ²)	Final (µg/g OC ²)		
1,2-DCB	23	100	1127	Not measured	289	1127
1,4-DCB	31	500	6121	9273	1345	6121
1,2,3-TCB	60	100	3455	Not measured	898	3455
1,2,4,5-TeCB	25	30	1582	Not measured	254	1582

¹ The lowest exposure concentration associated with mortality that is significantly different from acetone controls at 95% confidence level.

² % OC = 0.55%.

³ NOEC = No-Observed-Effect Concentration; LOEC = Lowest-Observed-Effect Concentration.

Table 7. Lowest effect concentration estimates in sediment ($\mu\text{g/g OC}$) based on effects data for water column organisms, calculated using the EqP method

CBz	K_{ow} ¹	Freshwater				Marine			
		Endpoint	Dissolved concentration ($\mu\text{g/L}$)	Reference	Estimated C_{sed} ($\mu\text{g/g OC}$)	Endpoint	Dissolved concentration ($\mu\text{g/L}$)	Reference	Estimated C_{sed} ($\mu\text{g/g OC}$)
1,2-DCB	2512	14-day EC_{50} reproduction (<i>Daphnia</i>)	550	Calamari <i>et al.</i> (1983)	1382	96-hour LC_{50} ² (mysid shrimp)	1970	U.S. EPA (1980a)	4949
1,4-DCB	2512	28-day LOEC reproduction (<i>Daphnia</i>)	400	Calamari <i>et al.</i> (1982)	1005	96-hour LC_{50} ² (mysid shrimp)	1990	U.S. EPA (1980a)	4999
TCBs	12 589	21-day LC_{90} (<i>Daphnia</i>)	130	Lay <i>et al.</i> (1985)	1637	reduced colonization of sediment (molluscs)	40	Tagatz <i>et al.</i> (1985)	504
TeCBs	31 623	16-day EC_{50} reproduction (<i>Daphnia</i>)	90	DeWolf <i>et al.</i> (1988)	2846	96-hour LC_{50} ² (mysid shrimp)	340	U.S. EPA (1980b)	10 752
QCB	100 000	16-day EC_{50} ² reproduction (<i>Daphnia</i>)	25	Hermens <i>et al.</i> (1984)	2500	96-hour LC_{50} ² (mysid shrimp)	160	US EPA (1980b)	16 000

¹ Mackay *et al.* (1992).

² Nominal concentration.

Table 8. Lowest effect concentrations ($\mu\text{g/g OC}$) for lettuce and earthworms

CBz	Species	Endpoint	Concentration ($\mu\text{g/g OC}$)	Source
1,2-DCB	–	no data	4712 ²	Assumed equal to 1,4-DCB
1,4-DCB	earthworm (<i>E. andrei</i>)	14-day LC ₅₀	4712 ²	van Gestel <i>et al.</i> (1991)
1,2,3-TCB	lettuce (<i>Lactuca sativa</i>)	14-day EC ₅₀ ¹ (growth)	119 ³	Hulzebos <i>et al.</i> (1993)
1,2,4-TCB	earthworm (<i>E. eugeniae</i>)	14-day LC ₅₀ ¹	2592 ²	Neuhauser <i>et al.</i> (1986)
1,3,5-TCB	lettuce	7-day EC ₅₀ ¹ (growth)	10 648 ⁴	Hulzebos <i>et al.</i> (1993)
1,2,3,4- TeCB	lettuce	14-day EC ₅₀ ¹ (growth)	2963 ⁴	Hulzebos <i>et al.</i> (1993)
1,2,4,5- TeCB	lettuce	14-day EC ₅₀ ¹ (growth)	185 ⁴	Hulzebos <i>et al.</i> (1993)
QCB	earthworm (<i>L. rubellus</i>)	14-day LC ₅₀	4136 ²	van Gestel <i>et al.</i> (1991)

¹ Determined using nominal concentrations.

² 4.9% OC.

³ 0.84% OC.

⁴ 1.08% OC.

Table 9. CTV_{SEDS} selected for benthic freshwater and marine organisms

CBz	Freshwater		Marine	
	CTV _{SED} (µg/g OC)	Data type	CTV _{SED} (µg/g OC)	Data type
1,2-DCB	1382	EqP-based chronic EC ₅₀	1127	Measured LOEC (<i>R. abronius</i>)
1,4-DCB	1005	EqP-based chronic LOEC	4999	EqP-based (acute effect)
TCBs	1637	EqP-based LC ₉₀	504	EqP-based (chronic effect)
TeCBs	2846	EqP-based chronic EC ₅₀	1582	Measured LOEC (<i>R. abronius</i>)
QCB	2500	EqP-based chronic EC ₅₀	3080	Median of extrapolated LOEC (<i>R. abronius</i>)

Table 10. Application factors and derived ENEV_{SEDS} for benthic organisms (freshwater and marine)

CBz	Freshwater				Marine				
	CTV _{SED} (µg/g OC)	Factors applied		ENEV _{SED} (µg/g OC)	CTV _{SED} (µg/g OC)	Factors applied			ENEV _{SED} (µg/g OC)
		chronic to no effects	P&B ¹			ACR	chronic to no effects	P&B ¹	
1,2-DCB	1382	10	–	138	1127	3	10	–	38
1,4-DCB	1005	10	–	101	4999	3	10	–	167
TCBs	1637	10	–	164	504	–	10	–	50
TeCBs	2846	10	10	29	1582	3	10	10	5
QCB	2500	10	10	25	3080	3	10	10	10

¹ P&B = persistence and bioaccumulation.

Table 11. Risk quotients for benthic organisms, based on maximum EEVs (EEV_{SEDS}) for Canadian sediments

CBz	Freshwater			Marine		
	Maximum EEV (µg/g OC)	ENEV (µg/g OC)	Quotient	Maximum EEV (µg/g OC)	ENEV (µg/g OC)	Quotient
1,2-DCB	52	138	0.4	<2.2	38	<0.06
1,4-DCB	522	101	5.2	40	167	0.24
TCBs	539	164	3.3	<2.2	50	<0.05
TeCBs	320	29	11	<2.2	5	<0.44
QCB	601	25	24	<2.2	10	<0.22

Table 12. Determination of ENEV_{SOILS} for soil-dwelling organisms

CBz	CTV _{SOIL} (µg/g OC)	Factor applied			ENEV _{SOIL} (µg/g OC)
		Limited data	Chronic to no effects	Persistence and bioaccumulation	
1,2-DCB	4712	3	10	–	157
1,4-DCB	4712	3	10	–	157
1,2,3-TCB	119	3	10	–	4.0
1,2,4-TCB	2592	3	10	–	86
1,2,3,4-TeCB	2963	3	10	10	9.9
1,2,4,5-TeCB	185	3	10	10	0.62
QCB	4136	3	10	10	14

Table 13. Risk quotients for terrestrial organisms, based on maximum EEVs (EEV_{SOILS}) for Canadian soils

CBz	Maximum EEV (µg/g OC)	Data source for maximum EEV	ENEV_{SOIL} (µg/g OC)	Quotient
1,2-DCB	0.42	Calculated from Webber and Nicols (1995)	157	0.003
1,4-DCB	0.87	Calculated from Webber and Nicols (1995)	157	0.006
1,2,3-TCB	0.05	Ding <i>et al.</i> (1992)	4.0	0.01
1,2,4-TCB	0.25	Ding <i>et al.</i> (1992)	86	0.003
1,2,3,4-TeCB	0.33	Ding <i>et al.</i> (1992)	9.9	0.03
1,2,4,5-/1,2,3,5-TeCB	0.40	Ding <i>et al.</i> (1992)	0.62	0.65
QCB	0.09	Ding <i>et al.</i> (1992)	14	0.006

Appendix 1. Persistence and bioaccumulation criteria as defined in the Persistence and Bioaccumulation Regulations of CEPA 1999

Persistence^a		Bioaccumulation^b
Medium	Half-life	
Air	≥ 2 days or is subject to atmospheric transport from its source to a remote area	BAF ^c ≥ 5000; BCF ^d ≥ 5000; log K _{ow} ^e ≥ 5
Water	≥ 182 days	
Sediment	≥ 365 days	
Soil	≥ 182 days	

^a A substance is persistent when at least one criterion is met in any one medium.

^b When the BAF of a substance cannot be determined in accordance with generally recognized methods, then the BCF of a substance will be considered, however, if neither its BAF nor its BCF can be determined with recognized methods, then the log K_{ow} will be considered.

^c Bioaccumulation factor means the ratio of the concentration of a substance in an organism to the concentration in water, based on uptake directly from the surrounding medium and food.

^d Bioconcentration factor means the ratio of the concentration of a substance in an organism to the concentration in water, based only on uptake directly from the surrounding medium.

^e octanol-water partition coefficient means the ratio of the concentration of a substance in an octanol phase to the concentration of the substance in the water phase of an octanol-water mixture

Appendix 2. Search Strategy - New Information for the Assessment of "Toxic" to the environment under Paragraph 64 (a) of CEPA 1999

To identify relevant information on Canadian production, importation, use, and environmental release, searches of the NPRI (National Pollutant Release Inventory, Environment Canada), the ARET (Accelerated Reduction/Elimination of Toxics, Environment Canada) were performed.

Data relevant to the assessment of whether 1,2-dichlorobenzene, 1,4-dichlorobenzene, trichlorobenzenes, tetrachlorobenzenes or pentachlorobenzene are "toxic" to the environment under paragraph 64 (a) of CEPA 1999 were identified from existing review documents, published reference checks and on-line searches of the following databases up to December, 1999. A search was conducted by name or CAS registry number in the following databases: Aquire, Registry of toxic effects of chemical substances (RTECS), Environment Abstracts, CAB abstracts, Current Contents, Poltox, Capulus Bib Abstracts, UnCover.



This report contains the collective views of an international group of experts and does not necessarily represent the decisions or the stated policy of the United Nations Environment Programme, the International Labour Organization, or the World Health Organization.

Concise International Chemical Assessment Document 60

**CHLOROENZENES OTHER THAN HEXACHLOROENZENE:
ENVIRONMENTAL ASPECTS**

First draft prepared by H.M. Malcolm, P.D. Howe, and S. Dobson, Centre for Ecology

and Hydrology, Monks Wood, United Kingdom

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World Health Organization

Geneva, 2004

The **International Programme on Chemical Safety (IPCS)**, established in 1980, is a joint venture of the United Nations Environment Programme (UNEP), the International Labour Organization (ILO), and the World Health Organization (WHO). The overall objectives of the IPCS are to establish the scientific basis for assessment of the risk to human health and the environment from exposure to chemicals, through international peer review processes, as a prerequisite for the promotion of chemical safety, and to provide technical assistance in strengthening national capacities for the sound management of chemicals.

The **Inter-Organization Programme for the Sound Management of Chemicals (IOMC)** was established in 1995 by UNEP, ILO, the Food and Agriculture Organization of the United Nations, WHO, the United Nations Industrial Development Organization, the United Nations Institute for Training and Research, and the Organisation for Economic Co-operation and Development (Participating Organizations), following recommendations made by the 1992 UN Conference on Environment and Development to strengthen cooperation and increase coordination in the field of chemical safety. The purpose of the IOMC is to promote coordination of the policies and activities pursued by the Participating Organizations, jointly or separately, to achieve the sound management of chemicals in relation to human health and the environment.

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FOREWORD

Concise International Chemical Assessment Documents (CICADs) are the latest in a family of publications from the International Programme on Chemical Safety (IPCS) — a cooperative programme of the World Health Organization (WHO), the International Labour Organization (ILO), and the United Nations Environment Programme (UNEP). CICADs join the Environmental Health Criteria documents (EHCs) as authoritative documents on the risk assessment of chemicals.

International Chemical Safety Cards on the relevant chemical(s) are attached at the end of the CICAD, to provide the reader with concise information on the protection of human health and on emergency action. They are produced in a separate peer-reviewed procedure at IPCS. They may be complemented by information from IPCS Poison Information Monographs (PIM), similarly produced separately from the CICAD process.

CICADs are concise documents that provide summaries of the relevant scientific information concerning the potential effects of chemicals upon human health and/or the environment. They are usually based on selected national or regional evaluation documents or on existing EHCs. Before acceptance for publication as CICADs by IPCS, these documents undergo extensive peer review by internationally selected experts to ensure their completeness, accuracy in the way in which the original data are represented, and the validity of the conclusions drawn.

The primary objective of CICADs is characterization of hazard and dose–response from exposure to a chemical. CICADs are not a summary of all available

data on a particular chemical; rather, they include only that information considered critical for characterization of the risk posed by the chemical. The critical studies are, however, presented in sufficient detail to support the conclusions drawn. For additional information, the reader should consult the identified source documents upon which the CICAD has been based.

Risks to human health and the environment will vary considerably depending upon the type and extent of exposure. Responsible authorities are strongly encouraged to characterize risk on the basis of locally measured or predicted exposure scenarios. To assist the reader, examples of exposure estimation and risk characterization are provided in CICADs, whenever possible. These examples cannot be considered as representing all possible exposure situations, but are provided as guidance only. The reader is referred to EHC 170.⁴

While every effort is made to ensure that CICADs represent the current status of knowledge, new information is being developed constantly. Unless otherwise stated, CICADs are based on a search of the scientific literature to the date shown in the executive summary. In the event that a reader becomes aware of new information that would change the conclusions drawn in a CICAD, the reader is requested to contact IPCS to inform it of the new information.

Procedures

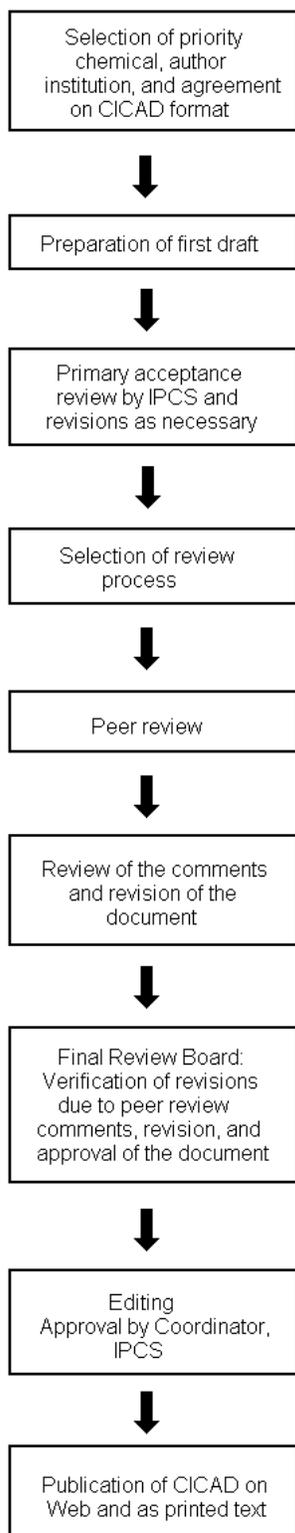
The flow chart on page 2 shows the procedures followed to produce a CICAD. These procedures are designed to take advantage of the expertise that exists around the world — expertise that is required to produce the high-quality evaluations of toxicological, exposure, and other data that are necessary for assessing risks to human health and/or the environment. The IPCS Risk Assessment Steering Group advises the Coordinator, IPCS, on the selection of chemicals for an IPCS risk assessment based on the following criteria:

- there is the probability of exposure; and/or
- there is significant toxicity/ecotoxicity.

Thus, it is typical of a priority chemical that

- it is of transboundary concern;
- it is of concern to a range of countries (developed, developing, and those with economies in transition) for possible risk management;
- there is significant international trade;
- it has high production volume;
- it has dispersive use.

CICAD PREPARATION FLOW CHART



Advice from Risk Assessment Steering Group

Criteria of priority:

- there is the probability of exposure; and/or
- there is significant toxicity/
- ecotoxicity.

Thus, it is typical of a priority chemical that

- it is of transboundary concern;
- it is of concern to a range of countries (developed, developing, and those with economies in transition) for possible risk management;
- there is significant international trade;
- the production volume is high;
- the use is dispersive.

Special emphasis is placed on avoiding duplication of effort by WHO and other international organizations.

A prerequisite of the production of a CICAD is the availability of a recent high-quality national/regional risk assessment document = source document. The source document and the CICAD may be produced in parallel. If the source document does not contain an environmental section, this may be produced *de novo*, provided it is not controversial. If no source document is available, IPCS may produce a *de novo* risk assessment document if the cost is justified.

Depending on the complexity and extent of controversy of the issues involved, the steering group may advise on different levels of peer review:

- standard IPCS Contact Points
- above + specialized experts
- above + consultative group

The Steering Group will also advise IPCS on the appropriate form of the document (i.e., a standard CICAD or a *de novo* CICAD) and which institution bears the responsibility of the document production, as well as on the type and extent of the international peer review.

The first draft is usually based on an existing national, regional, or international review. When no appropriate source document is available, a CICAD may be produced *de novo*. Authors of the first draft are usually, but not necessarily, from the institution that developed the original review. A standard outline has been developed to encourage consistency in form. The first draft undergoes primary review by IPCS to ensure that it meets the specified criteria for CICADs.

The second stage involves international peer review by scientists known for their particular expertise and by scientists selected from an international roster compiled by IPCS through recommendations from IPCS national Contact Points and from IPCS Participating Institutions. Adequate time is allowed for the selected experts to undertake a thorough review. Authors are required to take reviewers' comments into account and revise their draft, if necessary. The resulting second draft is submitted to a Final Review Board together with the reviewers' comments. At any stage in the international review process, a consultative group may be necessary to address specific areas of the science. When a CICAD is prepared *de novo*, a consultative group is normally convened.

The CICAD Final Review Board has several important functions:

- to ensure that each CICAD has been subjected to an appropriate and thorough peer review;
- to verify that the peer reviewers' comments have been addressed appropriately;
- to provide guidance to those responsible for the preparation of CICADs on how to resolve any remaining issues if, in the opinion of the Board, the author has not adequately addressed all comments of the reviewers; and
- to approve CICADs as international assessments.

Board members serve in their personal capacity, not as representatives of any organization, government, or industry. They are selected because of their expertise in human and environmental toxicology or because of their experience in the regulation of chemicals. Boards are chosen according to the range of expertise required for a meeting and the need for balanced geographic representation.

Board members, authors, reviewers, consultants, and advisers who participate in the preparation of a CICAD are required to declare any real or potential conflict of interest in relation to the subjects under discussion at any stage of the process. Representatives of nongovernmental organizations may be invited to observe the proceedings of the Final Review Board. Observers may participate in Board discussions only at the invitation of the Chairperson, and they may not participate in the final decision-making process.

1. EXECUTIVE SUMMARY

This CICAD on chlorobenzenes other than hexachlorobenzene (environmental aspects) is an update of *Environmental Health Criteria (EHC) 128, Chlorobenzenes other than hexachlorobenzene* (IPCS, 1991a). Information on the fate and levels of chlorobenzenes was also obtained from Agency for Toxic Substances and Disease Registry reports on chlorobenzene (ATSDR, 1990) and 1,4-dichlorobenzene (ATSDR, 1998). A further literature search was performed up to December 2002 to identify any additional information published since these reviews were completed. Information on the peer review of the source document is presented in Appendix 1. Information on the peer review of this CICAD is presented in Appendix 2. This CICAD was approved as an international assessment at a meeting of the Final Review Board, held in Varna, Bulgaria, on 8–11 September 2003. Participants at the Final Review Board meeting are listed in Appendix 3. The International Chemical Safety Cards for a number of different chlorobenzenes (ICSC 0037, 0344, 0531, 0642, 0676, 1049, 1066, 1095, 1222), produced by the International Programme on Chemical Safety (IPCS, 2000, 2003a–h), have also been reproduced in this document. This CICAD concentrates on environmental aspects because there have been no significant changes to the human health assessment since publication of the EHC (IPCS, 1991a).

Chlorinated benzenes are a group of cyclic aromatic compounds in which one or more hydrogen atoms of the benzene ring have been replaced by a chlorine atom. Chlorobenzenes are used mainly as intermediates in the synthesis of pesticides and other chemicals. 1,4-Dichlorobenzene (1,4-DCB) is used in space deodorants and as a moth repellent. The higher chlorinated benzenes (trichlorobenzenes, 1,2,3,4-tetrachlorobenzene [1,2,3,4-TeCB], and pentachlorobenzene [PeCB]) have been used as components of dielectric fluids.

Natural sources of chlorobenzenes in the environment have not been identified. Chlorobenzenes are released to the environment during manufacture or use as intermediates in the production of other chemicals. They will also be released during the disposal of chlorobenzene products, such as from incinerators and hazardous waste sites. Monochlorobenzene (MCB) is released directly to the environment due to its use as a pesticide carrier. Chlorobenzenes used as deodorizers, fumigants, degreasers, insecticides, herbicides, and defoliant will also be released to the environment as a direct result of their application.

Their physicochemical properties suggest that chlorobenzenes released to the environment are likely to volatilize to the atmosphere. Removal of chlorobenzenes from the atmosphere will occur primarily via reactions with hydroxyl radicals to produce nitrochlorobenzene, chlorophenol, and aliphatic dicarbonyl products, which are further removed by photolysis or reaction with hydroxyl radicals. Chlorobenzenes released into the aquatic environment will be redistributed preferentially to the air and to sediment (particularly organically rich sediments). Chlorobenzenes in aqueous solutions could, in theory, undergo photochemical reductive dechlorination, although studies have been performed only under artificial conditions that were not representative of temperate regions. The most important factor affecting the behaviour and fate of chlorobenzenes in soil is sorption. Adsorption-desorption processes in soil affect the rate of volatilization and leaching and the availability of chemicals to microbial and chemical degradation or uptake by plants or other organisms.

Chlorobenzenes in various substrates, including soil, sediment, and sewage sludge, may be degraded by microorganisms. The major mechanism of aerobic degradation is via oxidative dechlorination, leading to the formation of hydroxylated aromatic compounds (mainly catechols), which undergo ring fission and subsequent mineralization to carbon dioxide and water. The less chlorinated benzenes are more readily degraded than the higher chlorinated ones.

The bioaccumulation of chlorobenzenes by aquatic organisms is determined by their relative water and lipid solubilities (thus reflecting the octanol/water partition coefficients) and the number of chlorine substitutions. Uptake from water increases with increasing chlorination and increasing temperature.

Concentrations of chlorobenzenes (MCB, dichlorobenzenes, and trichlorobenzenes) have been reported in ambient air, with mean concentrations in the order of 0.1 µg/m³ and maximum levels (at hazardous waste sites) of up to 100 µg/m³. Concentrations of chlorobenzenes in surface waters are generally in the ng/litre to µg/litre range, with maximum concentrations up to 0.2 mg/litre in areas close to industrial sources. Levels of chlorobenzenes in industrial wastewaters may be higher and vary according to the nature of the processes used. Chlorobenzene levels in uncontaminated soils are generally less than 0.4 mg/kg for dichlorobenzene congeners and less than 0.1 mg/kg for other chlorobenzene congeners. Levels of chlorobenzenes in sediments are generally in the ng/kg to µg/kg range, although levels in the mg/kg range have been reported in samples from industrial areas.

In general, aquatic toxicity increases with the degree of chlorination of the benzene ring. Seventy-two-hour EC₅₀s for green algae range from 5280 µg/litre for 1,3-DCB to 200 000 µg/litre for MCB; similarly, 48-h EC₅₀s for diatoms range from 8 to 235 000 µg/litre. For freshwater invertebrates, 48-h EC₅₀s range from 10 µg/litre for PeCB to >530 000 µg/litre for 1,2,4,5-TeCB. Ninety-six-hour LC₅₀s for fish range from 135 µg/litre for PeCB to 21 000 µg/litre for 1,2,4-trichlorobenzene (1,2,4-TCB). Chronic no-observed-effect concentrations (NOECs) for freshwater invertebrates range from 32 µg/litre for PeCB to 19 000 µg/litre for MCB; in fish, NOECs range from 18 µg/litre for PeCB to 8500 µg/litre for MCB.

Few data are available on the effects of chlorobenzenes on terrestrial systems. LC₅₀ values for plants grown hydroponically or in soil ranged from 0.028 to 9.3 mg/litre and from 1 to >1000 mg/kg soil, respectively. LC₅₀ values for the earthworms *Eisenia andrei* and *Lumbricus rubellus* ranged from 0.22 µmol/litre (pore water) for PeCB to 4281 µmol/litre for MCB.

The risk of chlorinated benzenes causing harm to aquatic organisms is low. Risk factors comparing chronic toxicity values with concentrations measured in the environment were generally below 1, with the exception of some compounds that had higher risk factors, with a maximum value of 200. The highest risk factors were derived using old data from point sources and are therefore unrepresentative of the whole environment, especially when the likelihood of evaporation is considered. There were inadequate data to perform a risk assessment for terrestrial species.

2. IDENTITY AND PHYSICAL/CHEMICAL PROPERTIES

Chlorinated benzenes are a group of cyclic aromatic compounds in which one or more hydrogen atoms of the benzene ring have been replaced by a chlorine atom. The generic molecular formula is C₆H_{6-n}Cl_n, where n = 1–6. There are 12 different chlorinated benzenes: monochlorobenzene (MCB), dichlorobenzene (DCB) (three isomers), trichlorobenzene (TCB) (three isomers), tetrachlorobenzene (TeCB) (three isomers), pentachlorobenzene (PeCB), and hexachlorobenzene. Hexachlorobenzene is reviewed in a separate EHC (IPCS, 1997) and is therefore not covered by this CICAD.

The identity of chlorobenzenes and their physical and chemical properties are presented in Table 1. MCB, 1,2-DCB, 1,3-DCB, and 1,2,4-TCB are colourless liquids, while all other congeners are white crystalline solids at room temperature. In general, the solubility of chlorobenzenes in water is low (decreasing with increasing chlorination), flammability is low, the octanol/water partition coefficients are moderate to high (increasing with increasing chlorination), and vapour pressures are low to moderate (decreasing with increasing chlorination) (IPCS, 1991a).

Table 1: Physicochemical properties of chlorobenzenes.^a

Chlorinated benzene	Abbreviation	CAS No.	Molecular formula	Relative molecular mass	Melting point (°C)	Boiling point ^b (°C)	Vapour pressure at 25 °C (Pa)	Aqueous solubility at 25 °C (mg/litre)	Henry's law constant (kPa·m ³ /mol)	Log octanol/water partition coefficient (K _{ow})	Soil sorption coefficient (K _{oc})
Monochlorobenzene	MCB	108-90-7	C ₆ H ₅ Cl	112.6	-45.6	132.0	1665	293	0.377	2.98	466
1,2-Dichlorobenzene	1,2-DCB	95-50-1	C ₆ H ₄ Cl ₂	147.0	-17.0	180.5	197	91.1	0.198	3.38	987
1,3-Dichlorobenzene	1,3-DCB	541-73-1	C ₆ H ₄ Cl ₂	147.0	-24.7	173.0	269	123	0.366	3.48	1070

1,4-Dichlorobenzene	1,4-DCB	106-46-7	C ₆ H ₄ Cl ₂	147.0	53.1	174.0	90	30.9	0.160	3.38	1470
1,2,3-Trichlorobenzene	1,2,3-TCB	87-61-6	C ₆ H ₃ Cl ₃	181.5	53.5	218.5	17.3	12.2	0.306	4.04	3680
1,2,4-Trichlorobenzene	1,2,4-TCB	120-82-1	C ₆ H ₃ Cl ₃	181.5	17.0	213.5	45.3	45.3	0.439	3.98	2670
1,3,5-Trichlorobenzene	1,3,5-TCB	108-70-3	C ₆ H ₃ Cl ₃	181.5	63.5	208.0	24.0	3.99	0.233	4.02	NA ^c
1,2,3,4-Tetrachlorobenzene	1,2,3,4-TeCB	634-66-2	C ₆ H ₂ Cl ₄	215.9	47.5	254.0	5.2	12.1	0.261	4.55	NA
1,2,3,5-Tetrachlorobenzene	1,2,3,5-TeCB	634-90-2	C ₆ H ₂ Cl ₄	215.9	54.5	246.0	9.8	2.81	0.593	4.65	8560
1,2,4,5-Tetrachlorobenzene	1,2,4,5-TeCB	95-94-3	C ₆ H ₂ Cl ₄	215.9	139.5	243.6	0.72	2.16	0.261	4.51	6990
Pentachlorobenzene	PeCB	608-93-5	C ₆ HCl ₅	250.3	86.0	277.0	133 ^d	0.83	0.977	5.03	58 700

a From IPCS (1991a).

b Calculated at atmospheric pressure (101.3 kPa), except for 1,3,5-TCB, which was at 93.5 kPa.

c NA = not available.

d Calculated at 98 °C.

3. ANALYTICAL METHODS

The analytical technique of choice for the determination of chlorobenzenes in environmental samples is gas chromatography (GC). However, the methods of collection and preparation of samples for GC analysis vary considerably, depending on the medium and the laboratory. Capillary columns with different stationary phases are frequently used to separate compounds. Detection occurs via the use of a flame ionization detector (FID), electron capture detector (ECD), or mass spectrometric (MS) detector (IPCS, 1991a).

Tenax-GC resins have commonly been used as adsorbents for the air sampling of chlorobenzenes (Krost et al., 1982; Pellizzari et al., 1982), although XAD resins have also been used (Langhorst & Nestruck, 1979). Air pollutants collected on Tenax-GC resins can be desorbed directly onto the GC column by heating the tube with sorbent. XAD resins can be extracted with solvents, an aliquot of which can then be injected into a GC. Detection limits in the 1970s ranged from 0.7 µg/m³ for MCB to 0.9 µg/m³ for PeCB (Langhorst & Nestruck, 1979); however, much lower detection limits have been achieved more recently using ECD (0.5 pg/m³ for PeCB to 1.8 pg/m³ for 1,2,4,5-TeCB) (Hermanson et al., 1997).

Solvent extraction is a simple and effective technique for recovering chlorobenzenes from water samples. Hexane, pentane, and a 1:1 mixture of cyclohexane and diethyl ether have been identified as suitable extraction solvents for these compounds (Oliver & Bothen, 1980; Piet et al., 1980; Otson & Williams, 1981; Meharg et al., 2000). Alternatively, preconcentration of the chlorobenzenes on organic resins, such as Chromosorb 102 and Tenax-GC, is also effective; detection limits using Chromosorb 102 were reported to range from 0.5 µg/litre for MCB to 0.01 ng/litre for PeCB (Oliver & Bothen, 1980; Pankow & Isabelle, 1982). The purge-and-trap method has also been used to concentrate the volatile halogenated benzenes before analysis using GC (Jungclaus et al., 1978; Pereira & Hughes, 1980; Otson & Williams, 1982; Huybrechts et al., 2000; Martinez et al., 2002). Detection limits of 0.1–0.2 µg/litre for MCB and various dichlorobenzene isomers were achieved using FID and Hall electrolyte conductivity detectors (Otson & Williams, 1982), 0.08 µg/litre for 1,2,4-TCB using ECD (Martinez et al., 2002), and 0.76–20 ng/litre for di- and trichlorobenzenes using MS (Huybrechts et al., 2000). More recently, alternative extraction techniques such as headspace solid-phase microextraction with GC-MS have achieved detection limits for individual chlorobenzene isomers ranging from 4 to 6 ng/litre (He et al., 2000); however, it should be noted that analytical techniques using simple solvent extraction and GC-MS can now attain detection limits ranging from 5 pg/litre for 1,2,3- and 1,3,5-TCB to 15 pg/litre for PeCB (Meharg et al., 2000).

The extraction of chlorobenzenes from aquatic sediments, sewage sludges, or soil can be achieved by solvent or Soxhlet extraction (Oliver & Bothen, 1982; Lopez-Avila et al., 1983; Onuska & Terry, 1985; Wang & Jones, 1991; Wang et al., 1992). Solvents commonly used are acetone and/or hexane. Other extraction methods, such as sonication, saponification, and supercritical fluid extraction, have been used to extract sediment-bound chlorobenzenes, but were found to be less efficient than Soxhlet extraction (Prytula & Pavlostathis, 1996). The extract is generally dried using sodium sulfate, followed by cleanup on a Florisil column before GC analysis with ECD, with detection limits of 1500 µg/kg for MCB and lower detection limits ranging from 1.5 µg/kg for dichlorobenzenes to 0.05 µg/kg for PeCB (Oliver & Bothen, 1982; Onuska & Terry, 1985; Wang & Jones, 1991; Wang et al., 1992). Alternatively, headspace solid-phase microextraction with GC-ion trap MS has been found to reproduce detection limits of 0.03–0.1 µg/kg for 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB in soil (Santos et al., 1997).

For the detection of chlorobenzenes in biota samples, solvent or Soxhlet extraction with subsequent cleanup on Florisil columns and GC analysis with ECD have commonly been used (Lunde & Ofstad, 1976; Kuehl et al., 1980; Oliver & Bothen, 1982; Muir et al., 1992; Gebauer & Weseloh, 1993; Cobb et al., 1994; Jan et al., 1994; Wade et al., 1998). Detection limits of 1500 µg/kg for MCB and lower detection limits ranging from 5 µg/kg for dichlorobenzenes to 0.02 µg/kg for PeCB have been reported (Oliver & Bothen, 1982; Cobb et al., 1994). Vacuum extraction and the direct purge-and-trap method have also been used to quantify levels of MCB in fish tissue (Hiatt, 1981).

4. SOURCES OF ENVIRONMENTAL EXPOSURE

4.1 Natural sources

Natural sources of chlorobenzenes in the environment have not been identified. However, 1,2,3,4-TeCB has been identified in the oil of marsh grass, although it is not known whether this was formed naturally (Miles et al., 1973).

4.2 Anthropogenic sources

Chlorobenzenes are released to the environment from sites where they are either manufactured or used as intermediates in the production of other

chemicals. They will also be released during the disposal of chlorobenzene products, such as from incinerators (IPCS, 1991a) and hazardous waste sites (ATSDR, 1998). Chlorobenzenes are a product of incomplete combustion and may therefore be released to the environment from waste incinerators. Chlorobenzenes may be formed from the metabolic breakdown of lindane in higher organisms and from its physical breakdown under extreme environmental conditions (IPCS, 1991b).

Releases of some chlorobenzene compounds to the environment in the USA in 2001, as recorded in the US Toxics Release Inventory (TRI), are listed in Table 2. These data do not form a comprehensive list, as only certain types of industrial facility are required to register in the TRI (ATSDR, 1998). There is a paucity of data on the quantity of chlorobenzenes released to the environment in other parts of the world, although some production and consumption data are available. Approximately 15 000 tonnes of 1,4-DCB were produced in and/or imported into the European Union in 1994 (EC, 2001). Total production of MCB, 1,2-DCB, and 1,4-DCB in Japan in 1998 was 26 351 tonnes (Chemical Daily Company, 1999), with 9073 tonnes imported in 1998 and 8310 tonnes imported in 1999 (Chemical Daily Company, 2000).

Table 2: Total releases of chlorobenzenes in the USA during 2001.^a

	Releases (tonnes)					
	MCB	1,2-DCB	1,3-DCB	1,4-DCB	1,2,4-TCB	PeCB
Total emissions to air	314	56	0.50	37	43.92	0.03
Surface water discharges	0.3	0.38	0.26	0.51	0.04	0.06
Releases to land	0.01	0.00	0.00	0.00	3.5	1.07
Total on-site releases	362	59	0.76	42	49	1.16
Total off-site releases	2.5	0.52	0.46	0.69	4.2	0.09

^a From US EPA (2003).

Some uses of chlorobenzenes, including uses as deodorizers, fumigants, degreasers, insecticides, herbicides, and defoliant, will result in direct releases to the environment.

MCB will be released directly to the environment due to its use as a pesticide carrier (Meek et al., 1994c). MCB is used as a solvent carrier for pesticides (29 000 kg per annum in Canada), in the manufacture of rubber polymers (20 000 kg per annum in Canada), and as a carrier for textile dyes (1000 kg per annum in Canada) (Mackay et al., 1996). Fifty per cent of the MCB used in Canada is released to the environment; 80% is emitted to the atmosphere, 10% to water, and 10% to soil, giving releases of 20 000, 2500, and 2500 tonnes, respectively, per year (Mackay et al., 1996). MCB is used in the production of phenol and nitrochlorobenzene (*ortho* and *para* isomers), in the formulation of herbicides, to produce additional chlorobenzenes, and as a solvent in the manufacture of adhesives, paints, resins, dyestuffs, and drugs (Grosjean, 1991). MCB is used in the manufacture of diphenyl oxide, phenylphenol, silicone resin, and other halogenated organics (ATSDR, 1990).

1,2-DCB is used primarily in the automotive and metal industries as a solvent for the removal of carbon and degreasing of metal parts (Meek et al., 1994a). 1,2-DCB is used in the synthesis of organic chemicals such as toluene diisocyanate (Grosjean, 1991).

1,4-DCB is used in air fresheners, urinal deodorants, and moth and bird repellents (Meek et al., 1994b; EC, 2001). All of these uses release 1,4-DCB to the environment, principally the atmosphere. 1,4-DCB is also used as an intermediate in the production of other chemicals, including polyphenylene sulfide resins (Grosjean, 1991) and 1,2,4-TCB (ATSDR, 1998). Minor uses of 1,4-DCB include its use in the control of tree-boring insects, ants, and blue mould in tobacco seedbeds (ATSDR, 1998).

Trichlorobenzenes, especially 1,2,4-TCB, are used as dye carriers, degreasing solvents, oil additives, and dielectric fluids and in the formulation of pesticides (Grosjean, 1991). The use of trichlorobenzenes is restricted to mainly 1,2,4-TCB, which is used as a chemical intermediate and an industrial solvent (Giddings et al., 1994c). 1,2,4-TCB was formerly used as a degreasing agent, in septic tanks, and in drain cleaners, wood preservatives, and abrasive formulations (EC, 2003).

Tetrachlorobenzenes and pentachlorobenzenes may be released to the environment from the spillage of dielectric fluids (Giddings et al., 1994a,b). 1,2,3,4-TeCB is used as a component in dielectric fluids (IPCS, 1991a). 1,2,4,5-TeCB is used as an intermediate in the manufacture of herbicides and defoliant. It is also used as an insecticide, as a moisture-resistant impregnant, in electrical insulation, and in packing protection (IPCS, 1991a). PeCB was formerly used in a pesticide to combat oyster drills (small snails that eat oysters). It has also been used as an intermediate (IPCS, 1991a).

5. ENVIRONMENTAL TRANSPORT, DISTRIBUTION, AND TRANSFORMATION

5.1 Transport and distribution

Their physicochemical properties suggest that chlorobenzenes released to the environment are likely to be volatilized to the atmosphere. The Henry's law constants measured for chlorobenzenes suggest that they are readily volatilized, especially from aquatic systems with long residence times, such as large lakes and oceans (Ten Hulscher et al., 1992). However, chlorobenzenes released to water may also be adsorbed onto sediment, especially if it is rich in organic matter. Volatilization from soil is also likely, although, depending on the characteristics of the soil, there may also be sorption to soil.

The majority of chlorobenzenes added to soil, as either sewage sludge or spiked samples, were volatilized, with biodegradation and abiotic degradation insignificant compared with the amount volatilized (Wang & Jones, 1994a). Volatilization occurred by two-step first-order processes, with high rates of volatilization during an initial step, followed by a second, much slower step, which was presumably controlled by the rate of desorption of the compound from soil. Half-lives for loss of chlorobenzenes ranged from 13.0 to 219 days for sewage sludge applications and from 10.6 to 103 days for spiked samples. Half-lives increased with increasing chlorination and were also higher in sludge-amended soil than in the spiked samples. The half-lives for volatilization of MCB and 1,2-DCB from soil were 2.1 and 4.0 days, respectively. Initial soil concentrations were 100 mg/kg dry weight (Anderson et al., 1991). Transient

geochemical conditions can significantly alter the extent of removal. Robertson (1994) studied the fate of a dichlorobenzene mixture (containing 74% 1,2-DCB, 11% 1,3-DCB, and 15% 1,4-DCB) released to sub-surface soil in effluent from a septic system. High dichlorobenzene concentrations were found in the aerobic unsaturated zone (below the septic system) where dichlorobenzene had a residence time of 60 days. The migration of dichlorobenzene to the water table was attenuated by this zone.

The most important factor affecting the behaviour and fate of chlorobenzenes in soil is sorption. Adsorption-desorption processes in soil affect the rate of volatilization and leaching and the availability of chemicals to microbial and chemical degradation or uptake by plants or other organisms (Wang & Jones, 1994a). The soil sorption coefficients for chlorobenzenes range from 466 to 58 700 (Table 1) and generally increase with increasing chlorination (IPCS, 1991a; Schrap et al., 1994). Sorption of chlorobenzenes to soil is affected by many parameters, and it increases with increasing organic matter content (Barber et al., 1992; Faschan et al., 1993).

The adsorption of 1,2,4-TCB to soil was found to decrease with increasing soil depth (Njoroge et al., 1998). These depth-related changes were attributed to changes in composition, texture, and accessibility of the soil organic matter. At deeper levels, extractable organic matter was increasingly dominated by fulvic acids. The higher fulvic:humic acid ratio in deep soil reflects an increasing hydrophilicity of the soil organic matter. Abundance of iron oxide and size of clay particles also increase with depth.

Sorption of chlorobenzenes is also affected by soil moisture, with reduced sorption to wet soil (Chiou & Shoup, 1985; Thibaud et al., 1993). Adsorption of 1,2,4-TCB to soil was reduced following the addition of sodium dodecyl sulfate, a surfactant that frequently occurs in sewage sludge disposed of onto land (DiVincenzo & Dentel, 1996). Desorption occurred only when the sodium dodecyl sulfate concentration exceeded the critical micelle concentration. However, increased adsorption of MCB to soil was reported following the addition of the surfactant hexadecyltrimethylammonium (HDTMA) (Sheng et al., 1998). Adsorption on the HDTMA phase was 80–160 times higher than sorption on natural organic matter. The sorption of 1,4-DCB by aquifer materials with a low organic carbon content was enhanced in the presence of tetrachloroethene (Brusseau, 1991). The enhanced sorption was suggested to arise from tetrachloroethene increasing the organic carbon content of the sorbent.

Desorption of 1,3-DCB from a silty soil to deionized water had an initial fast labile phase, followed by a slow phase (Lee et al., 2002). An average of 60% of the initial concentration was desorbed. The first-order rate constant was 0.022–0.038 per hour for the labile phase and 4.1×10^{-5} to 7.8×10^{-4} per hour for the slow desorption phase. Single-step batch tests showed that desorption of chlorobenzenes from sediment was slow, with less than 0.5% of 1,2,4,5-TeCB and PeCB desorbed within 62 days. Desorption of 1,2,4-TCB was significantly higher than that of other compounds, with 3% desorbed within 62 days (Gess & Pavlostathis, 1997).

MCB adsorbed onto marine sediment reached equilibrium within 3 h (Zhao et al., 2001). Equilibrium took the same time in natural seawater, artificial seawater, and deionized water. Adsorption occurred via the surface and micropores of sediment and could be described by either the Freundlich or Langmuir model. Adsorption was not affected by temperature (18, 25, or 30 °C), although the saturate adsorption amount decreased at higher temperatures. Adsorption isotherms and the saturate adsorption amounts were higher in natural seawater than in artificial seawater and deionized water. Adsorption of 1,2,4,5-TeCB on sandy aquifer solids took up to hundreds of days to reach equilibrium (Ball & Roberts, 1991). Distribution coefficients were greatest in the size fraction with the largest grains.

Mean (\pm SD) suspended sediment/water partition coefficients ($\log K_{oc}$) for chlorobenzenes measured in Ise Bay, Japan, were 3.47 ± 0.74 (1,3-DCB), 3.69 ± 0.48 (1,2-DCB), 3.61 ± 0.39 (1,2,3-TCB), 3.86 ± 0.40 (1,2,4-TCB), 3.55 ± 0.47 (1,3,5-TCB), 4.39 ± 0.33 (1,2,3,4-TeCB), 3.94 ± 0.33 (1,2,3,5-TeCB and 1,2,4,5-TeCB), and 4.59 ± 0.41 (PeCB) (Masunaga et al., 1996b). Concentrations of chlorobenzenes in water and adsorbed onto suspended sediment were compared. None of the chlorobenzenes gave a clear adsorbed level distribution pattern, and the correlation between soluble and adsorbed chlorobenzenes was weak.

The fate of MCB, 1,2-DCB, and 1,2,4-TCB in wastewater applied to soil was examined in a microcosm experiment (Piwoni et al., 1986). Initial concentrations of MCB, 1,2-DCB, and 1,2,4-TCB in the wastewater were 1.9–3.1, 2.4, and 0.72 $\mu\text{mol/litre}$, respectively. The proportions of MCB and 1,2-DCB volatilized were 14% and 21%, respectively, and it was assumed that 84% and 79%, respectively, were degraded, giving concentrations in the volume effluent of $9 \pm 10\%$ of the original concentration. Volatility of 1,2,4-TCB was not measured, but it was assumed to be approximately 89%, as $<0.7\%$ of the original concentration remained in the effluent.

The half-life for dichlorobenzene (all isomers) in a septic groundwater system was 15 days (Robertson, 1994). The site included a 2-m-thick, sandy aerobic unsaturated zone. This loss was due to a combination of volatilization and aerobic biodegradation. Biodegradation occurred after an initial lag phase and was most likely for 1,3-DCB and 1,4-DCB. Dichlorobenzene in the anaerobic zone was not readily biodegraded.

Octanol/air partition coefficients ($\log K_{oa}$) measured for chlorobenzenes at 25 °C were 4.36 (1,2-DCB), 5.19 (1,2,3-TCB), 5.64 (1,2,3,4-TeCB), 5.63 (1,2,4,5-TeCB), and 6.27 (PeCB) (Harner & Mackay, 1995). Octanol/air partition coefficients determined partitioning from the atmosphere to vegetation, soils, and possibly aerosols.

Microcosm experiments suggested that 1,2-DCB in soil was not taken up by grass (*Holcus lanatus*) roots, although some foliar adsorption of dichlorobenzene volatilized from soil was reported (Wilson & Meharg, 1999). A root concentration factor of 19 litres/kg has been reported for 1,2,4-TCB (Dietz & Schnoor, 2001). From these data, it can be assumed that tri- and/or tetrachlorinated benzenes have the potential to be taken up by plants.

5.2 Transformation

5.2.1 Abiotic degradation

Removal of chlorobenzenes from the atmosphere will occur primarily via reactions with hydroxyl radicals to produce nitrochlorobenzene, chlorophenol, and aliphatic dicarbonyl products, which are further removed by photolysis or reaction with hydroxyl radicals. Photolysis and reactions with ozone or nitrate radicals are of negligible importance (Grosjean, 1991). Rate constants for reactions with hydroxyl radicals (in cm^3/s per molecule) were calculated to be 8.8×10^{-13} (MCB), 4.0×10^{-13} (1,2-DCB), 7.2×10^{-13} (1,3-DCB), 4.3×10^{-13} (1,4-DCB), 6.0×10^{-13} (1,2,3-TCB), and 5.65×10^{-13} (1,2,4-TCB) (Atkinson et al., 1985; Klöpffer et al., 1986; Dilling et al., 1988; Arnts et al., 1989). A rate constant for reaction of MCB with ozone was calculated to be $<5 \times 10^{-21}$ cm^3/s per molecule. Assuming 24-h average hydroxyl radical and ozone concentrations of 1×10^6 and 7.2×10^{11} molecules/ cm^3 , tropospheric half-lives for MCB were calculated to be 13 days for reactions with hydroxyl radicals and >8.8 years for reactions with ozone (Atkinson et al., 1985). Tropospheric half-lives for 1,4-DCB and 1,2,4-TCB reacting with hydroxyl radicals were calculated to be 33.4 and 26.7 days, respectively (Klöpffer et al., 1988).

1,2,4-TCB in the atmosphere may be degraded via direct photolysis, although this route of degradation is minor, due to the poor spectral overlap between the solar spectrum and the adsorption spectrum of 1,2,4-TCB. The maximum photolysis rate for 1,2,4-TCB in summer at midday under clear skies was 0.03% per hour (Bunce et al., 1989).

Chlorobenzenes in aqueous solutions may undergo photochemical reductive dechlorination. PeCB was degraded to tetrachlorobenzenes, which in turn were photodegraded to trichlorobenzenes, dichlorobenzenes, MCB, and, ultimately, phenol, benzene, and hydrogen chloride (Chu & Jafvert, 1994). These reactions were reported following exposure to 253.7-nm monochromatic ultraviolet lamps. The rate of photodegradation increased in the presence of surfactants. In addition to the main reductive pathway of photodechlorination, minor pathways, including photochlorination, photohydrolysis, and photoisomerization, also occurred. 1,2,3,5-TeCB was photolysed to 1,2,4-TCB or 1,3,5-TCB in the presence of an acetone sensitizer (Choudhry & Hutzinger, 1984). Photochemical reactions in the absence of a sensitizer transformed tetrachlorobenzenes into other isomers and also produced some chlorobenzenes with greater chlorination than the original tetrachlorobenzene compound. The rate constant for reaction of 1,2,4-TCB with hydroxyl radicals in an acidic solution was $6.0 \pm 0.3 \times 10^9$ per mol/litre per second (Gallard & De Laat, 2001). 1,4-DCB in aqueous solution was photodegraded to 4-chlorophenol, hydroquinone, hydroxybenzoquinone, and 2,5-dichlorophenol (Meunier et al., 2001). The formation of 2,5-dichlorophenol demonstrates hydroxylation without dechlorination. Photodegradation of MCB in aqueous solutions has been reported under both aerobic and anaerobic conditions and at pHs ranging from <1 to <12 (Tissot et al., 1983, 1984; Dilmeghani & Zahir, 2001). Degradation followed first-order kinetics, with rate constants ranging from 1.8×10^{-4} to 6.4×10^{-4} per second for anaerobic and oxygen-saturated conditions, respectively (Dilmeghani & Zahir, 2001). The rate of degradation was an order of magnitude higher with ultraviolet and hydrogen peroxide or hydrogen peroxide–ozone compared with ultraviolet alone.

The half-lives for photolytic degradation of MCB and 1,2,4-TCB in surface water, simulating summer conditions at 40 degrees latitude, were 170 and 450 years, respectively (Dulin et al., 1986).

5.2.2 Biodegradation

Chlorobenzenes in various substrates, including soil, sediment, and sewage sludge, can be degraded by microorganisms. The major mechanism of aerobic degradation is via oxidative dechlorination, usually initiated by dioxygenative hydroxylation, leading to the formation of hydroxylated aromatic compounds (mainly catechols), which undergo ring fission and subsequent mineralization to carbon dioxide and water. The less chlorinated benzenes are more readily degraded than the higher chlorinated ones (IPCS, 1991a). Biodegradation under anaerobic conditions has also been reported, although this occurs at a slower rate than aerobic biodegradation.

Chlorobenzene-degrading bacteria isolated from aerobic environments include *Burkholderia* (previously known as *Pseudomonas*) species (strains JS150, P51, JS6, PS12, and PS14) (Pettigrew et al., 1991; Sander et al., 1991, 1997; Van der Meer et al., 1991, 1997; Nishino et al., 1994; Beil et al., 1997; Meckenstock et al., 1998), *Alcaligenes* species (strains A175 and OBB65) (De Bont et al., 1986; Schraa et al., 1986), *Escherichia hermannii* (Kiernicka et al., 1999), *Nitrosomonas europaea* (Keener & Arp, 1994), *Mycobacterium vaccae*, and *Rhodococcus* species (strain R22) (Fairlee et al., 1997).

The degradative abilities of these bacteria vary, with some organisms exhibiting a lag or adaptation period prior to degradation. Some can degrade several chlorobenzenes (Brunsbach & Reineke, 1994), whereas others are compound-specific (Reineke & Knackmuss, 1984; Brunsbach & Reineke, 1994; Keener & Arp, 1994). For some, degradation occurs only in the presence of other sources of carbon and energy, whereas others are able to use chlorobenzenes as their sole carbon and energy source (Van der Meer et al., 1987). Genetic analysis has shown that these bacteria contain a novel combination of previously existing genes — genes for aromatic ring dioxygenase and dihydrodiol dehydrogenase — and other genes for a chlorocatechol oxidative pathway.

Degradation is also dependent upon the initial chlorinated benzene concentrations. Degradation will occur only if the initial concentration is below the toxic threshold, although bacteria that have previously been exposed to MCB have the ability to degrade higher concentrations than those that did not have prior exposure. For example, concentrations of MCB greater than 2.5 mmol/litre (282 mg/litre) were found to be toxic to *Pseudomonas* sp. strain RHO1 cells. Cells that had previously been exposed to MCB demonstrated toxicity at concentrations greater than 3.5 mmol/litre (394 mg/litre) (Fritz et al., 1992).

MCB and 1,2,4-TCB were degraded by bacteria isolated from solids sampled from pristine aquifers (Swindoll et al., 1988). Degradation followed first-order rate constants, with V_{\max} values of 0.38–2.71 ng/g per hour for 1,2,4-TCB. Degradation of MCB was not saturated; therefore, V_{\max} could not be calculated.

A consortium of Gram-negative and Gram-positive bacteria isolated from groundwater and soil contaminated with MCB was able to mineralize 54% of a 2.23 $\mu\text{mol/litre}$ solution via the modified *ortho* pathway within 7 days in the presence of nutrients. Degradation also occurred without added nutrients, although at a slower rate (Nishino et al., 1992).

Degradation of 1,4-DCB occurred at similar rates under aerobic or anaerobic conditions and was enhanced in mixtures with high sludge content (which reduced overall oxygen) (Gejlsbjerg et al., 2001). Mineralization occurred after a lag phase of 30 days. After inoculation for 2 months, mineralization was 12.4% in the sludge and 21.6% in the 1:20 sludge:soil mixture. The authors concluded that mineralization was probably occurring in the aerobic layers of the sludge–soil mixtures, as mineralization did not occur in sludge in the absence of molecular oxygen.

A consortium of bacteria isolated from Rhine sediment was able to degrade PeCB, 1,2,3,4-TeCB, 1,2,3,5-TeCB, 1,2,4,5-TeCB, and 1,2,3-TCB via reductive dechlorination in the presence of lactate, glucose, ethanol, or isopropanol as the electron donor (Holliger et al., 1992). PeCB was degraded to 1,3,5-TCB, while 1,2,3,4-TeCB and 1,2,4,5-TeCB were degraded to 1,2,4-TCB. Chlorobenzenes that were not dechlorinated during the 4-week incubation included 1,2,4-TCB, 1,3,5-TCB, and all isomers of dichlorobenzene. Other studies have reported complete mineralization of some higher chlorinated compounds. Two *Pseudomonas* strains (PS12 and PS14) isolated from the soil of an industrial waste deposit were able to mineralize various chlorobenzenes, including MCB, all three dichlorobenzenes, 1,2,4-TCB, and 1,2,4,5-TeCB (strain PS14 only). 1,2,4-TCB and 1,2,4,5-TeCB were degraded via dioxygenation of the aromatic ring, producing 3,4,6-trichlorocatechol. Subsequent *ortho* cleavage, catalysed by a Type II catechol 1,2-dioxygenase, produced 2,3,5-trichloromuconate, which was degraded via the tricarboxylic acid pathway (Sander et al., 1991).

Degradation of 1,2-DCB and 1,4-DCB within a mixture of organic compounds was reported in a 149-day batch microcosm using sediment and groundwater obtained from various sampling sites of an aquifer (Nielsen & Christensen, 1994). The initial concentrations were approximately 120 $\mu\text{g/litre}$. Within an average of 82 days, 78.3% of 1,4-DCB and 81.0% of 1,2-DCB were degraded. The lag phases were 4.9 and 4.5 days for 1,4-DCB and 1,2-DCB, respectively. In the Organisation for Economic Co-operation and Development (OECD) closed bottle test, 67% of an initial 1,4-DCB concentration of 1.9 mg/litre was mineralized after 28 days, indicating that 1,4-DCB is readily degradable (Topping, 1987).

Bartholomew & Pfaender (1983) calculated degradation rates for MCB and 1,2,4-TCB at different sites of a river system during different seasons. Rates of degradation of MCB and 1,2,4-TCB were reported to decrease over the freshwater to estuarine to marine gradient. V_{\max} values for MCB degradation during

May and September were 13–14 ng/litre per hour for fresh water, 4.9–10 ng/litre per hour for estuarine water, and <1–1.7 ng/litre per hour for marine water. V_{\max} values were <1 ng/litre per hour at all three sites in February. The corresponding values for degradation of 1,2,4-TCB in May and July were <1–7.5 ng/litre per hour for fresh water, <1–7.9 ng/litre per hour for estuarine water, and <1–2.3 ng/litre per hour for marine water.

In controlled lysimeter experiments, 80% of 1,2,4,5-TeCB in soils and liquid cultures was mineralized by the bacterial strains *Isphingomonas* sp. strains HH69 and RW1 and *Pseudomonas* sp. strain PS14 within a few days (Figge et al., 1993). Degradation was not increased in the presence of additional energy sources such as peptone, triolein, and glucose. Degradation did not occur in acidic soils (pH < 4).

Biodegradation of chlorobenzenes has also been reported in several studies under anaerobic conditions, including methanogenic and sulfate-reducing conditions. As with aerobic degradation, degradability varies between organisms. Under anaerobic conditions, degradation is limited to dechlorination, with no breakdown of the aromatic structure.

Anaerobic degradation of chlorobenzenes has been reported in river sediment (Masunaga et al., 1996a; Susarla et al., 1996). Dechlorination occurred without a lag period, with half-lives ranging from 17 to 433 days. The main pathway for PeCB dechlorination was via 1,2,4,5-TeCB, 1,2,4-TCB, 1,4-DCB, and MCB. A minor pathway, via 1,2,3,4-TeCB, 1,2,3-TCB, 1,2-DCB, and MCB, was also observed. MCB was stable under anaerobic conditions. The preferences for dechlorination were two adjacent chlorine atoms, followed by one chlorine on an adjacent carbon, followed by no chlorine on the adjacent carbon. Other studies have reported similar anaerobic biodegradation (Beurskens et al., 1991; Ramanand et al., 1993; Susarla et al., 1997). Nowak et al. (1996) reported anaerobic degradation of all chlorobenzenes, including MCB, to benzene.

In anaerobic sewage sludge, PeCB was dechlorinated to 1,2,3,4-TeCB and 1,2,3,5-TeCB, which were degraded to 1,2,4-TCB, 1,2,3-TCB, and 1,3,5-TCB, and then 1,2-DCB and 1,3-DCB (Yuan et al., 1999). Sequential dechlorination occurred within a substrate concentration range of 2–50 mg/litre, but was slower at concentrations greater than 50 mg/litre. Dechlorination rates were highest under methanogenic conditions (0.30 mg/litre per day), with slower rates under sulfate-reducing (0.12 mg/litre per day) and denitrifying conditions (0.08 mg/litre per day). The rate of dechlorination of 1,2,3-TCB by anaerobic sediment ranged from 15 to 35 pmol/ml wet sediment per day (Yonezawa et al., 1994).

Some studies have shown chlorobenzenes to be resistant to anaerobic biodegradation. Nielsen et al. (1995) reported no biodegradation of 1,2-DCB or 1,4-DCB in anaerobic landfill leachate collected from four different sites at distances ranging from 2 to 350 m from the landfill. The governing reactions, which varied at each site, included methanogenesis, iron(III) reduction, nitrate reduction, and manganese(IV) reduction. Dichlorobenzenes have been reported to persist for at least 20 years in an aquifer that had been contaminated with rapid-infiltration sewage disposal (Barber, 1988). 1,2,3,5-TeCB and 1,3,5-TCB were resistant to degradation by soil slurry microorganisms that could degrade PeCB, 1,2,3,4-TeCB, and 1,2,4-TCB (Ramanand et al., 1993).

5.3 Bioaccumulation

The bioaccumulation of chlorobenzenes by aquatic organisms is determined by their relative water and lipid solubility (thus reflecting the octanol/water partition coefficients) and the number of chlorine substitutions. Uptake from water increases with increasing chlorination (Könemann & Van Leeuwen, 1980; Oliver & Niimi, 1983; Sabljic, 1987; Koelmans & Jimenez, 1994; Wang et al., 1997) and with increasing temperature (Koelmans & Jimenez, 1994).

Mean bioconcentration factors (BCFs) (dry weight) for phytoplankton increased from 4700 for 1,2,3-TCB at 4.5 °C to 26 000 for PeCB at 38.6 °C (Koelmans & Jimenez, 1994). Wang et al. (1997) found significant differences in the accumulation of chlorobenzenes by different marine algal species, with BCFs (dry weight) ranging from 600 to 3000 for 1,2,3,4-TeCB and from 1000 to 6000 for PeCB.

BCFs ranging from 270 for 1,2-DCB to 20 000 for PeCB were reported for laboratory studies on rainbow trout (*Oncorhynchus mykiss*) (Oliver & Niimi, 1983). BCFs for a variety of fish species ranged from 7000 to 24 000 (lipid weight) for 1,2,4-TCB, with a positive correlation between bioaccumulation and lipid content (Geyer et al., 1985). Galassi & Calamari (1983) found BCFs (lipid weight) ranging from 4000 to 22 000 for 1,2,3- and 1,2,4-TCB in rainbow trout, with newly hatched fish accumulating 2–4 times the amount found in eyed eggs or young fish (alevins). Qiao et al. (2000) report that gill uptake of 1,2,4-TCB and PeCB could account for 98% of the body burden. Uptake of trichlorobenzenes, tetrachlorobenzenes, and PeCB was significantly reduced by the presence of suspended particles (Schrapp & Opperhuizen, 1990). However, PeCB was found to be readily desorbed from sediments with a low organic carbon content and subsequently accumulated by fish via the gills (Qiao & Farrell, 1996). The rate of elimination of chlorobenzenes decreases with increasing chlorination (Melancon & Lech, 1985; De Boer et al., 1994). Elimination half-lives for dichlorobenzenes to PeCB in laboratory-exposed fish ranged from 0.05 to 1.6 days (Melancon & Lech, 1985). However, for eels (*Anguilla anguilla*) transferred from a contaminated lake to a "clean" lake, elimination half-lives of >300 days were reported for tetrachlorobenzenes and PeCB (De Boer et al., 1994). Sijm & Van der Linde (1995) calculated elimination rate constants and predicted elimination half-lives for 1,2,3-TCB to be 40 days in small fish, such as guppies (*Poecilia reticulata*), and >5 years in larger and/or fatty fish.

The coefficient of adsorption onto sediment influences the uptake into terrestrial plants and sediment-living aquatic invertebrates; the degree of chlorination is also correlated with uptake (Knezovich & Harrison, 1988; IPCS, 1991a). Under non-equilibrium conditions, BCFs for chironomid midge larvae exposed to sediment-bound chlorobenzenes were 5, 29, and 225 for MCB, 1,2-DCB, and 1,2,4-TCB, respectively. BCFs were best correlated with the concentrations of the chlorobenzenes in the interstitial water (Knezovich & Harrison, 1988).

The tri- and tetrachlorinated benzenes may be taken up by plants, as indicated by the root concentration factor of 19 litres/kg reported for 1,2,4-TCB (Dietz & Schnoor, 2001).

However, the prediction of BCFs is more difficult for terrestrial plants than for aquatic organisms because of the complex nature of the root soil interface combined with gaseous uptake by aerial parts (Scheunert et al., 1994). Topp et al. (1986) compared the uptake of chlorobenzenes by plants from the soil and via the air in closed, aerated laboratory systems. A negative correlation was demonstrated between the BCF and the soil adsorption coefficient (based on soil organic matter content) for the uptake into the roots of barley. The adsorption of chlorobenzenes onto soil organic matter increased with increasing chlorination. However, expression of uptake in barley roots in relation to the soil interstitial water concentration of the chlorobenzenes produced a positive correlation between the BCF and the octanol/water partition coefficients. Higher chlorinated chlorobenzenes, therefore, are most readily taken up by the plant roots when they are available in soil interstitial water. This will occur particularly in sandy soils with low organic matter content. In a later study, Topp et al. (1989) found that after growth in soil containing 2 µg each of 1,2,4-TCB and PeCB per kg dry weight, harvested barley grain contained 73 and 82 µg/plant, respectively. The concentrations in the dry grain were 0.05 and 0.06 mg/kg for 1,2,4-TCB and PeCB, respectively. In further studies on soybeans (*Glycine max*), linear correlations were found between equilibrium tissue/water coefficients, the octanol/water partition coefficient, and measured lipid content (Tam et al., 1996). The bioconcentration of chlorobenzenes into excised soybean (*Glycine max*) roots increased exponentially with increasing octanol/water partition coefficient (Kraaij & Connell, 1997). Wang & Jones (1994b) concluded that the total amount of chlorobenzenes taken up by carrots grown in sewage sludge-amended and spiked soils was low (<1%) compared with other loss pathways from the soil, principally volatilization.

Belfroid et al. (1994) calculated BCFs for earthworms (*Eisenia andrei*) of 104 and 156 for 1,2,3,4-TeCB and PeCB in soil; BCFs based on interstitial water were 67 000 and 307 000, respectively, and were found to be similar to BCFs found for worms exposed in water alone (Belfroid et al., 1993). BCFs for earthworms exposed via water show a clear increase in uptake of chlorobenzenes with increasing chlorination, and steady-state concentrations are reached within 5 days (Belfroid et al., 1993). Elimination rate constants reveal that chlorobenzene loss decreases with increasing chlorination. A monophasic elimination curve was observed in water, whereas biphasic elimination was found in the presence of soil (Belfroid et al., 1993); elimination rates in soil experiments were significantly increased by the addition of organic matter (Belfroid & Sijm, 1998). Feeding studies have revealed that earthworms can also take up chlorobenzenes via food. In studies with field-contaminated soil, steady-state concentrations in worms were much lower than in laboratory studies, suggesting decreased bioavailability of chlorobenzenes (Belfroid et al., 1995).

6. ENVIRONMENTAL LEVELS

Chlorobenzene (MCB, dichlorobenzenes, and trichlorobenzenes) concentrations have previously been reported in ambient air, with mean concentrations in the order of 0.1 $\mu\text{g}/\text{m}^3$ and maximum levels of up to 100 $\mu\text{g}/\text{m}^3$ at hazardous waste sites (IPCS, 1991a). Popp et al. (2000) measured tetrachlorobenzenes and PeCB in air sampled from two industrially contaminated sites and a reference site in Germany in 1998. Mean gas-phase concentrations of tetrachlorobenzenes and PeCB at the contaminated sites ranged from 5.7 to 30.9 pg/m^3 and from 10.2 to 28 pg/m^3 , respectively. Mean concentrations at the control site ranged from 6.4 to 10.6 pg/m^3 . Particulate-bound chlorobenzenes accounted for 1.9% of the total concentrations. A low proportion of particulate-bound chlorobenzenes was also reported in air sampled from the Bering and Chukchi seas in 1993 (Strachan et al., 2001). Mean gas-phase concentrations for the Bering Sea were 1.1, 4.0, and 6.6 pg/m^3 for 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB, respectively, and for the Chukchi Sea, 2.8, 10, and 14 pg/m^3 , respectively. Mean chlorobenzene concentrations at four sites throughout Michigan, USA (1992–1994), ranged from 22 to 30 pg/m^3 for 1,2,4,5-TeCB, from 40 to 53 pg/m^3 for 1,2,3,4-TeCB, and from 35 to 69 pg/m^3 for PeCB (Hermanson et al., 1997). Annual mean concentrations for southern Ontario, Canada (1988–1989), were $>5.3 \text{ pg}/\text{m}^3$ for 1,2,3,4-TeCB and $>8.0 \text{ pg}/\text{m}^3$ for PeCB (Hoff et al., 1992). Higher concentrations have been reported in close proximity to pollution sources. A concentration of 5 $\mu\text{g}/\text{m}^3$ for tri- and tetrachlorobenzenes was found within 200 m of an electro-industrial plant in Slovenia (Jan et al., 1994). Seasonal variations in the concentrations of 1,4-DCB in ambient air have also been reported, with concentrations increasing with increasing temperature (Hanai et al., 1985).

Chlorobenzenes have also been detected in rainwater, their presence presumably being due to transfer from the ambient air. Concentrations of all three dichlorobenzene isomers and 1,2,4-TCB in rainwater were less than 10 ng/litre at selected sites in Oregon and California, USA (Pankow et al., 1983). In the United Kingdom, 1,4-DCB was detected in rainwater at a mean concentration of $10 \pm 5 \text{ ng}/\text{litre}$ (Fielding et al., 1981).

In 12 sewage sludges in the United Kingdom, the concentrations of chlorobenzenes ranged from $<0.01 \text{ mg}/\text{kg}$ dry weight for PeCB to 40.2 mg/kg dry weight for 1,3-DCB, with a general reduction in concentration with increased chlorine substitution (Rogers et al., 1989). Further sampling of United Kingdom sewage sludges revealed chlorobenzene concentrations ranging from 35 100 to 192 000 mg/kg dry weight for MCB, from 13 to 4110 mg/kg for dichlorobenzenes, from 2 to 1070 mg/kg for trichlorobenzenes, from 0.2 to 101 mg/kg for tetrachlorobenzenes, and from 2 to 37 mg/kg for PeCB (Wang & Jones, 1994c). Analysis of archived sludge samples showed that concentrations of 1,4-DCB increased over the period 1942–1961, whereas other chlorobenzenes increased in concentration only from 1954 onwards (Wang et al., 1992).

Data on levels of the lower chlorinated benzenes (MCB, dichlorobenzenes, and trichlorobenzenes) in wastewater indicate that MCB is detected the most often and at the highest concentrations, occasionally exceeding 1 mg/litre . Chlorobenzene concentrations in US wastewater have been reported to range from 11 to 6400 $\mu\text{g}/\text{litre}$ for MCB, from 10 to 860 $\mu\text{g}/\text{litre}$ for dichlorobenzenes, and from 12 to 607 $\mu\text{g}/\text{litre}$ for trichlorobenzenes (IPCS, 1991a).

Concentrations of chlorobenzenes in surface waters are generally in the ng/litre to $\mu\text{g}/\text{litre}$ range, with maximum concentrations up to 0.2 mg/litre in areas close to industrial sources (IPCS, 1991a). Mean concentrations of dissolved chlorobenzenes in the Bering and Chukchi seas ranged from 3 to 10 pg/litre for 1,2,3-TCB, from 15 to 36 pg/litre for 1,2,3,4-TeCB, and from 9 to 36 pg/litre for PeCB (Strachan et al., 2001). Higher chlorobenzene levels have been detected in coastal waters and estuaries, with Dutch coastal waters containing mean concentrations ranging from 9 to 117 ng/litre for dichlorobenzenes and from 0.7 to 1.6 ng/litre for trichlorobenzenes (Van de Meent et al., 1986) and Japanese coastal waters containing mean dissolved concentrations ranging from 24.3 ng/litre for 1,3-DCB to 0.25 ng/litre for tetrachlorobenzenes (Masunaga et al., 1996b). Waters of the Scheldt estuary (The Netherlands) contained chlorobenzene concentrations ranging from <130 to 315 ng/litre for dichlorobenzenes, from <25 to 320 ng/litre for trichlorobenzenes, and from <45 to 135 ng/litre for tetrachlorobenzenes (Van Zoest & Van Eck, 1991); more recent sampling revealed MCB concentrations ranging from 5 to 31.5 ng/litre (Huybrechts et al., 2000). Chlorobenzene concentrations of up to 500 ng/litre have been reported for MCB in the Tees Estuary, United Kingdom (Law et al., 1991), and for 1,3-DCB in Yokkaichi Port, Ise Bay, Japan, during 1988 (Masunaga et al., 1991a). Mean chlorobenzene concentrations in the Forth Estuary, United Kingdom, during 1987 ranged from <0.1 to 790 ng/litre for dichlorobenzenes, from 4 to 5500 ng/litre for trichlorobenzenes, from <0.04 to 20 ng/litre for tetrachlorobenzenes, and from <0.01 to 40 ng/litre for PeCB. The predominant isomers detected were 1,2,3- and 1,2,4-TCB, and these were found near industrial effluent discharges (Rogers et al., 1989; Harper et al., 1992). Further studies in 1990 revealed 1,2,3- and 1,2,4-TCB concentrations ranging up to 51 and 84 ng/litre , respectively (Harper et al., 1992).

The highest chlorobenzene concentrations in surface waters have been reported for river waters in heavily populated and/or industrialized areas. Mean concentrations in the river Besos, Spain, were 260 ng/litre for MCB, 600 ng/litre for 1,4-DCB, 5000 ng/litre for 1,2-DCB and 1,3-DCB, 1100 ng/litre for 1,2,3-TCB, and 8100 ng/litre for 1,2,4-TCB (Gomez Belinchon et al., 1991). Concentrations of MCB and 1,4-DCB ranging from non-detected to $>10 \mu\text{g}/\text{litre}$ have been reported for both compounds in water from the Ohio River (US EPA, 1985). Elder et al. (1981) reported trichlorobenzene concentrations (isomer not specified) ranging from 0.1 to 8 $\mu\text{g}/\text{litre}$ in water from Niagara Falls, New York, USA. Corresponding concentrations of tetrachlorobenzene ranged from 0.1 to 200 $\mu\text{g}/\text{litre}$. Concentrations of PeCB in water sampled from the Great Lakes ranged from not detected to 0.0006 $\mu\text{g}/\text{litre}$ (Oliver & Nicol, 1982). Concentrations in water sampled from the rivers and estuary of Osaka (a major urban area of Japan) ranged from 0.2 to 30 $\mu\text{g}/\text{litre}$ for MCB, from 0.17 to 130 $\mu\text{g}/\text{litre}$ for 1,4-DCB, from 0.2 to 10 $\mu\text{g}/\text{litre}$ for 1,2-DCB, from 0.16 to 0.35 $\mu\text{g}/\text{litre}$ for 1,2,4-TCB, and from 0.18 to 0.30 $\mu\text{g}/\text{litre}$ for 1,2,3-TCB (Yamamoto et al., 1997).

Mean chlorobenzene concentrations in sediment from the Bering and Chukchi seas ranged from 0.02 to 0.41 $\mu\text{g}/\text{kg}$ for 1,2,3-TCB, from 0.08 to 0.87 $\mu\text{g}/\text{kg}$ for 1,2,3,4-TeCB, and from 0.33 to 0.4 $\mu\text{g}/\text{kg}$ for PeCB (Strachan et al., 2001). Mean concentrations in coastal sediments from Ise Bay, Japan, were 4.8 $\mu\text{g}/\text{kg}$ for 1,2,4-TCB, 2.3 $\mu\text{g}/\text{kg}$ for 1,2-DCB, 1.9 $\mu\text{g}/\text{kg}$ for 1,3-DCB, and $<0.15 \mu\text{g}/\text{kg}$ for 1,3,5-TCB, tetrachlorobenzenes, and PeCB (Masunaga et al., 1991b). Lee & Fang (1997) reported mean values for the Tsen-wen estuary, Taiwan, of 3.2 $\mu\text{g}/\text{kg}$ for 1,2-DCB, 20.7 $\mu\text{g}/\text{kg}$ for 1,3-DCB, and 11.2 $\mu\text{g}/\text{kg}$ for 1,2,4-TCB.

Lake Garda (Italy) contained mean sediment PeCB concentrations of 0.2 $\mu\text{g}/\text{kg}$ dry weight (Bossi et al., 1992), whereas Lake Superior (Hamilton Harbour, Canada) contained levels ranging from 3.6 $\mu\text{g}/\text{kg}$ for PeCB to 80 $\mu\text{g}/\text{kg}$ for 1,4-DCB (Onuska & Terry, 1985). Sediment samples from the river Elbe,

Germany, ranged from 30 to 740 µg/kg dry weight for MCB, from 20 to 1060 µg/kg for dichlorobenzenes (1,2- and 1,4-DCB), from 1 to 115 µg/kg for trichlorobenzenes (1,2,3- and 1,2,4-TCB), from 1 to 27 µg/kg for tetrachlorobenzenes, and from 1 to 14 µg/kg for PeCB (Götz et al., 1993), whereas samples from the river Rhine contained concentrations ranging from 40 to 240 µg/kg dry weight for dichlorobenzenes, from <10 to 20 µg/kg for trichlorobenzenes, and from <0.5 to 2 µg/kg for PeCB (Alberti, 1983).

Chlorobenzene levels in uncontaminated soils are generally less than 0.4 mg/kg for dichlorobenzene congeners and less than 0.1 mg/kg for other chlorobenzene congeners (Wang et al., 1995). Multiple applications of sewage sludge can increase the chlorobenzene content in sludge-amended soil compared with control soils. However, Wang et al. (1995) found that most chlorobenzenes disappear rapidly on cessation of sludge application, with around 10% remaining 30 years later. They found that 1,4-DCB levels increased significantly in United Kingdom soils during the 1960s to a maximum mean value in 1967 of 10 mg/kg in control soils and 16.6 mg/kg in sludge-amended soils. Analysis of subsoil from a former pesticide factory in Germany showed that tetrachlorobenzenes and PeCB were dominant in the upper soil layers (up to 1.9 m), accounting for 80% of chlorobenzenes, with 1,2,3,4-TeCB and PeCB accounting for 44% and 24%, respectively. At depths between 1.9 and 5.5 m, trichlorobenzenes were more dominant, accounting for 60%, with 1,2,4-TCB accounting for 37% (Feidieker et al., 1994). Total chlorobenzene concentrations ranged from 1.5 to 18 400 mg/kg.

Mean chlorobenzene concentrations in bivalves from US coastal waters ranged from <0.25 to 28.2 µg/kg dry weight for 1,2,4,5-TeCB, from <0.25 to 10 µg/kg for 1,2,3,4-TeCB, and from <0.25 to 13.3 µg/kg for PeCB (Wade et al., 1998). Aquatic insects from a variety of Canadian sites contained mean PeCB concentrations ranging from <0.49 to 21.4 µg/kg dry weight (Ciborowski & Corkum, 1988). Concentrations in freshwater and marine fish from contaminated areas range from 0.1 to 50 µg/kg wet weight, with higher chlorinated compounds generally present at the highest concentrations (IPCS, 1991a). The eggs of fish-eating birds contained mean PeCB levels of 1.2 and 4.4 µg/kg from two sites in Puget Sound, USA (Cobb et al., 1994). Waterfowl from Lake Ontario, Canada, contained mean chlorobenzene concentrations ranging from 0.3 to 1.7 µg/kg wet weight for 1,2,3,4-TeCB and from 0.65 to 33.4 µg/kg for 1,2,4,5-TeCB (Gebauer & Weseloh, 1993). Mean concentrations in Arctic marine mammal blubber ranged from 1 to 9.7 µg/kg wet weight for 1,2,3,4-TeCB and from 16.8 to 20.2 µg/kg for PeCB (Muir et al., 1992; Weis & Muir, 1997).

7. EFFECTS ON ORGANISMS IN THE LABORATORY AND FIELD

7.1 Aquatic environment

The acute toxicity of chlorobenzenes to aquatic organisms is presented in Table 3. Forty-eight-hour EC₅₀s for diatoms range from 8 to 235 000 µg/litre. For freshwater invertebrates, 48-h EC₅₀s range from 10 µg/litre for PeCB to >530 000 µg/litre for 1,2,4,5-TeCB. Ninety-six-hour LC₅₀s for fish range from 135 for PeCB in the freshwater guppy (*Poecilia reticulata*) to 21 000 µg/litre for 1,2,4-TCB in the saltwater sheepshead minnow (*Cyprinodon variegatus*).

Table 3: Acute toxicity of chlorobenzenes to aquatic species.

Organism	End-point	Chlorobenzene	Test conditions ^a	Concentration (µg/litre)	Reference
Microorganisms — Saltwater					
Diatom (<i>Cyclotella meneghiniana</i>)	48-h EC ₅₀ (DNA measurement)	MCB	M	235 740	Figueroa & Simmons (1991)
		1,2-DCB	M	23 330	Figueroa & Simmons (1991)
		1,3-DCB	M	51 880	Figueroa & Simmons (1991)
		1,4-DCB	M	34 300	Figueroa & Simmons (1991)
		1,2,3-TCB	M	6420	Figueroa & Simmons (1991)
		1,2,4-TCB	M	2830	Figueroa & Simmons (1991)
		1,3,5-TCB	M	590	Figueroa & Simmons (1991)
		1,2,3,5-TeCB	M	1370	Figueroa & Simmons (1991)
		1,2,3,4-TeCB	M	1390	Figueroa & Simmons (1991)
		1,2,4,5-TeCB	M	270	Figueroa & Simmons (1991)
PeCB	M	8	Figueroa & Simmons (1991)		
Invertebrates — Freshwater					
Water flea (<i>Daphnia magna</i>)	48-h EC ₅₀ /LC ₅₀	MCB	C M	585.52	Rose et al. (1998)
		MCB	C N	12 900–17 300 ^b	Cowgill et al. (1985)
		MCB	C N	5810	Abernethy et al. (1986)

		MCB	C N	86 000	LeBlanc (1980)
		1,2-DCB	C N	2352	Abernethy et al. (1986)
		1,2-DCB	N	740–2200 ^c	Canton et al. (1985)
		1,2-DCB	C M	4200–7400	Richter et al. (1983)
		1,2-DCB	C N	2400	LeBlanc (1980)
		1,3-DCB	N	1200–6800 ^c	Canton et al. (1985)
		1,3-DCB	N	10 500–13 500	Gersich et al. (1986)
		1,3-DCB	C N	28 000	LeBlanc (1980)
		1,4-DCB	N	700–2200 ^c	Canton et al. (1985)
		1,4-DCB	C N	11 000	LeBlanc (1980)
		1,2,3-TCB	C N	1452	Abernethy et al. (1986)
		1,2,4-TCB	C M	1700–2100	Richter et al. (1983)
		1,2,4-TCB	C N	50 000	LeBlanc (1980)
		1,2,4,5-TeCB	C N	>530 000	LeBlanc (1980)
		PeCB	C N	300	Abernethy et al. (1986)
Water flea (<i>Ceriodaphnia dubia</i>)	48-h EC ₅₀ /LC ₅₀	MCB	M C	7900–47 000	Rose et al. (1998)
		MCB	C N	8900–11 100 ^d	Cowgill et al. (1985)
		1,2-DCB	M C	661.5	Rose et al. (1998)
		1,2,4-TCB	M C	308	Rose et al. (1998)
		1,2,3,4-TeCB	M C	130	Rose et al. (1998)
		PeCB	M C	10	Rose et al. (1998)
Midge (<i>Tanytarsus dissimilis</i>)	48-h LC ₅₀	1,2-DCB		2300–11 800	Call et al. (1979)
		1,4-DCB		13 000	Call et al. (1983)
Midge (<i>Chironomus thummi</i>)	48-h LC ₅₀	1,4-DCB	C M	1200	Roghair et al. (1994)
		1,2,3-TCB	C M	1700	Roghair et al. (1994)
		1,2,3,4-TeCB	C M	540–730	Roghair et al. (1994)
		PeCB	C M	230–320	Roghair et al. (1994)
Invertebrates — Saltwater					
Fleshy prawn (<i>Penaeus chinensis</i>)	96-h LC ₅₀	MCB		1720	Yin & Lu (1993)
Crab (<i>Portunus pelagicus</i>)	96-h EC ₅₀ (growth)	MCB	C N	748	Mortimer & Connell (1994)
		1,4-DCB	C N	201	Mortimer & Connell (1994)
		1,2,3-TCB	C N	173	Mortimer & Connell (1994)
		1,2,3,4-TeCB	C N	410	Mortimer & Connell (1994)
		PeCB	C N	87	Mortimer & Connell (1994)
Grass shrimp (<i>Palaemonetes pugio</i>)	96-h LC ₅₀	1,2-DCB	M	9400	Curtis et al. (1979)
		1,4-DCB	M	69 000	Curtis et al. (1979)
		1,2,4-TCB	M	540	Clark et al. (1987)
Opossum shrimp (<i>Americamysis</i>)	96-h LC ₅₀	1,3-DCB		2850	US EPA (1978)

bahia)		1,2,4-TCB		450	US EPA (1978)
		1,2,4,5-TeCB		1480	US EPA (1978)
		PeCB		160	US EPA (1978)
Fish — Freshwater					
Rainbow trout (<i>Oncorhynchus mykiss</i>)	96-h LC ₅₀	MCB	M	4700	Dalich et al. (1982)
		1,2-DCB		1520–1580	Call et al. (1979)
		1,4-DCB		880	Mayer & Ellersieck (1986)
		1,4-DCB		1120	Call et al. (1983)
		1,2,4-TCB		1530	Call et al. (1983)
		1,2,4,5-TeCB	N	1200–10 000 ^e	Van Leeuwen et al. (1985)
		PeCB		190	Call et al. (1979)
	16-day LC ₅₀	MCB	C N	90	Birge et al. (1979a)
14-day LC ₅₀	1,4-DCB	C M	800	Calamari et al. (1983)	
Fathead minnow (<i>Pimephales promelas</i>)	96-h LC ₅₀	MCB	C M	7700	Marchini et al. (1993)
		MCB	M	16 900	Geiger et al. (1990)
		1,2-DCB	M	6027	Sijm et al. (1993)
		1,3-DCB	M	7800	Carlson & Kosian (1987)
		1,4-DCB	M	4200	Carlson & Kosian (1987)
		1,2,4-TCB	M	2760	Carlson & Kosian (1987)
		1,2,4-TCB	M	2990	Geiger et al. (1990)
		1,2,3,4-TeCB	M	1100	Carlson & Kosian (1987)
		1,2,4,5-TeCB		89–460	Brooke (1991)
Goldfish (<i>Carassius auratus</i>)	96-h LC ₅₀	MCB	C N	2370–3480 ^f	Birge et al. (1979a)
Guppy (<i>Poecilia reticulata</i>)	96-h LC ₅₀	1,2,3-TCB	M	348	Van Hoogen & Opperhuizen (1988)
		1,2,3-TCB	M	365	Van Hoogen & Opperhuizen (1988)
		PeCB	M	135	Van Hoogen & Opperhuizen (1988)
	14-day LC ₅₀	MCB	C N	24 964	Könemann (1981)
		1,2-DCB	C N	5852	Könemann (1981)
		1,3-DCB	C N	7367	Könemann (1981)
		1,4-DCB	C N	3957	Könemann (1981)
		1,2,4-TCB	C N	2393	Könemann (1981)
		1,3,5-TCB	C N	3302	Könemann (1981)
	14-day LC ₅₀	1,2,4,5-TeCB	C N	305	Könemann (1981)
		PeCB	C N	177	Könemann (1981)
Mosquitofish (<i>Gambusia affinis</i>)	96-h LC ₅₀	1,2,3-TCB	C M	2196	Chaisuksant et al. (1998)
	96-h LC ₅₀	PeCB	C M	200	Chaisuksant et al. (1998)

Largemouth bass (<i>Micropterus salmoides</i>)	7.5-day LC ₅₀	MCB	C M	50–60 ^g	Birge et al. (1979b)
Zebrafish (<i>Brachydanio rerio</i>)	28-day LC ₅₀	MCB	M	10 300	Van Leeuwen et al. (1990)
	7- to 28-day NOEC	MCB	M	8500	Van Leeuwen et al. (1990)
	14- to 28-day NOEC	1,4-DCB	M	2100	Van Leeuwen et al. (1990)
	28-day LC ₅₀	1,2,3-TCB	M	990	Van Leeuwen et al. (1990)
	14- to 28-day NOEC	1,2,3-TCB	M	450	Van Leeuwen et al. (1990)
	14- to 28-day NOEC	1,2,4-TCB	M	450	Van Leeuwen et al. (1990)
	28-day LC ₅₀	1,2,3,4-TeCB	M	410	Van Leeuwen et al. (1990)
	7- to 21-day NOEC	1,2,3,4-TeCB	M	310	Van Leeuwen et al. (1990)
	7- to 28-day NOEC	PeCB	M	110	Van Leeuwen et al. (1990)
Fish — Saltwater					
Dover sole (<i>Solea solea</i>)	96-h LC ₅₀	MCB	C M	5821	Furay & Smith (1995)
European flounder (<i>Platichthys flesus</i>)	96-h LC ₅₀	MCB	C M	6609	Furay & Smith (1995)
		1,2-DCB	C M	4616	Furay & Smith (1995)
		1,2,4-TCB	C M	8585	Furay & Smith (1995)
Sheepshead minnow (<i>Cyprinodon variegatus</i>)	96-h LC ₅₀	MCB	N	10 000	Heitmuller et al. (1981)
		1,2-DCB	N	9700	Heitmuller et al. (1981)
		1,3-DCB	N	7800	Heitmuller et al. (1981)
		1,4-DCB	N	7400	Heitmuller et al. (1981)
		1,2,4-TCB	N	21 000	Heitmuller et al. (1981)
		1,2,3,5-TeCB	N	3700	Heitmuller et al. (1981)
		1,2,4,5-TeCB	N	800	Heitmuller et al. (1981)
		1,2,4,5-TeCB	M	330	Ward et al. (1981)
		PeCB	N	800	Heitmuller et al. (1981)
Sheepshead minnow (<i>Cyprinodon variegatus</i>)	28-day NOEC (growth)	PeCB		18–86 ^h	Hansen & Cripe (1991)
	28-day NOEC (survival)	PeCB		19–120 ^h	Hansen & Cripe (1991)

^a C = test carried out in a closed exposure system; M = measured exposure concentration; N = nominal exposure concentration.

^b Range of values for differences in temperature: 12 900 refers to 20 °C, 17 300 refers to 24 °C.

^c Range indicated difference between EC₅₀ (lower value) and LC₅₀ (upper value).

^d Range of values for differences in temperature: 8900 refers to 20 °C, 11 100 refers to 24 °C.

^e Range of values for different life stages: 10 000 refers to all egg stages and sac fry; 1200 refers to early fry.

^f Range of values for differences in water hardness: 2370 refers to 200 mg calcium carbonate/litre; 3480 refers to 50 mg calcium carbonate/litre.

^g Range of values for differences in water hardness: 50 refers to 50 mg calcium carbonate/litre; 60 refers to 200 mg calcium carbonate/litre.

^h Range of values due to interlaboratory comparisons.

Chronic toxicity of chlorobenzenes to aquatic organisms is presented in Table 4. Seventy-two-hour EC_{50} s for green algae range from 5280 $\mu\text{g/litre}$ for 1,3-DCB to 200 000 $\mu\text{g/litre}$ for MCB. Chronic NOECs for freshwater invertebrates range from 32 $\mu\text{g/litre}$ (PeCB) to 19 000 $\mu\text{g/litre}$ (MCB) for reproduction and from <1400 to 3890 $\mu\text{g/litre}$ (MCB) for survival. In fish, NOECs range from 18 $\mu\text{g/litre}$ for PeCB to 8500 $\mu\text{g/litre}$ for MCB. The data in this table include the standardized ecotoxicological tests, although any species shown to be more sensitive to the effects of chlorobenzenes have also been included. Care must be taken when interpreting these data, as chlorobenzenes may volatilize from the test system (with the lower chlorinated compounds generally more volatile), thus reducing actual exposure.

Table 4: Chronic toxicity of chlorobenzenes to aquatic organisms.

Organism	End-point	Chlorobenzene	Test conditions ^a	Concentration ($\mu\text{g/litre}$)	Reference
Microorganisms — Freshwater					
Green algae (<i>Pseudokirchneriella subcapitata</i>) ^b	72-h EC_{50} (population)	MCB	N	202 000	US EPA (1978)
		1,2-DCB	N	76 100	US EPA (1978)
		1,3-DCB	N	5280	US EPA (1978)
		1,4-DCB	N	77 500	US EPA (1978)
		1,2,4-TCB	N	21 700	US EPA (1978)
		1,2,3,5-TeCB	N	14 700	US EPA (1978)
		PeCB	N	13 000	US EPA (1978)
Invertebrates — Freshwater					
Water flea (<i>Daphnia magna</i>)	10-day LC_{50}	MCB	C N	16 000	Cowgill & Milazzo (1991)
	9- to 11-day NOEC (survival)	MCB	C N	<1400	Cowgill & Milazzo (1991)
	9- to 11-day EC_{50} (reproduction)	MCB	C N	15 000–19 000 ^c	Cowgill & Milazzo (1991)
	9- to 11-day NOEC (reproduction)	MCB	C N	6500–11 000 ^d	Cowgill & Milazzo (1991)
	Four-brood EC_{50} (reproduction)	MCB	M	1912	De Wolf et al. (1988)
	Four-brood NOEC (reproduction)	MCB	M	1004	De Wolf et al. (1988)
	14-day EC_{50} (reproduction)	1,2-DCB	C M	550	Calamari et al. (1983)
	16-day EC_{50} (reproduction)	1,3-DCB	M	1400	Hermens et al. (1984)
	14-day EC_{50} (reproduction)	1,4-DCB	C M	930	Calamari et al. (1983)
	14-day EC_{50} (reproduction)	1,2,3-TCB	C M	200	Calamari et al. (1983)
	Four-brood EC_{50} (reproduction)	1,2,4-TCB	M	330	De Wolf et al. (1988)
	Four-brood NOEC (reproduction)	1,2,4-TCB	M	182	De Wolf et al. (1988)
	14-day EC_{50} (reproduction)	1,2,4-TCB	C M	450	Calamari et al. (1983)
	Four-brood EC_{50} (reproduction)	1,2,3,4-TeCB	M	90	De Wolf et al. (1988)
	Four-brood NOEC (reproduction)	1,2,3,4-TeCB	M	55	De Wolf et al. (1988)
	Four-brood EC_{50} (reproduction)	PeCB	M	61	De Wolf et al. (1988)
Four-brood NOEC (reproduction)	PeCB	M	32	De Wolf et al. (1988)	

Water flea (<i>Ceriodaphnia dubia</i>)	7-day LC ₅₀	MCB	C N	24 000	Cowgill & Milazzo (1991)
	7- to 10-day NOEC (survival)	MCB	C N	3890	Cowgill & Milazzo (1991)
	7- to 10-day EC ₅₀ (reproduction)	MCB	C N	14 000–26 000 ^c	Cowgill & Milazzo (1991)
	7- to 10-day NOEC (reproduction)	MCB	C N	12 000–19 000 ^f	Cowgill & Milazzo (1991)
Invertebrates — Saltwater					
Crab (<i>Portunus pelagicus</i>)	40-day NOEC (growth)	MCB	C N	125	Mortimer & Connell (1995)
		1,4-DCB	C N	31	Mortimer & Connell (1995)
		1,2,3-TCB	C N	25	Mortimer & Connell (1995)
		1,2,3,4-TeCB	C N	17	Mortimer & Connell (1995)
		PeCB	C N	5	Mortimer & Connell (1995)
Fish — Freshwater					
Rainbow trout (<i>Oncorhynchus mykiss</i>)	85-day LOEC (growth)	1,2,4-TCB	M	36.3–90.75	Hodson et al. (1991)
	85-day LOEC (survival)	1,2,4-TCB	M	454–853	Hodson et al. (1991)
Fish — Saltwater					
Sheepshead minnow (<i>Cyprinodon variegatus</i>)	28-day NOEC (growth)	PeCB		18–86 ^g	Hansen & Cripe (1991)
	28-day NOEC (survival)	PeCB		19–120 ^g	Hansen & Cripe (1991)

a C = test carried out in a closed exposure system; M = measured exposure concentration; N = nominal exposure concentration.

b Previously known as *Selenastrum capricornutum*.

c Range of EC₅₀ values refers to different measurements of reproduction. EC₅₀ values for progeny, broods, and mean brood size were 15 000, 19 000, and 16 000 µg/litre, respectively.

d Range of NOEC values refers to different measurements of reproduction. NOEC values for progeny, broods, and mean brood size were 11 000, 11 000, and 6500 µg/litre, respectively.

e Range of EC₅₀ values refers to different measurements of reproduction. EC₅₀ values for progeny, broods, and mean brood size were 14 000, 26 000, and 22 000 µg/litre, respectively.

f Range of NOEC values refers to different measurements of reproduction. NOEC values for progeny, broods, and mean brood size were 19 000, 19 000, and 12 000 µg/litre, respectively.

g Range of values due to interlaboratory comparisons.

Concentrations of MCB greater than 2.5 mmol/litre (282 mg/litre) were found to be toxic to *Pseudomonas* sp. strain RHO1 cells. Cells that had previously been exposed to MCB demonstrated toxicity at concentrations greater than 3.5 mmol/litre (394 mg/litre) (Fritz et al., 1992). Figueroa & Simmons (1991) measured the effect of chlorobenzenes on diatom (*Cyclotella meneghiniana*) DNA. DNA quantification was used as a biological indicator of cellular biomass. Toxicity increased with increasing chlorination and was related to physicochemical and structural properties. Toxicity was explained in terms of partitioning between the aqueous and biological phases.

The acute toxicity of MCB to daphnids (*Daphnia magna* and *Ceriodaphnia dubia*) was higher at 20 °C than at 24 °C. Forty-eight-hour LC₅₀ values for *Daphnia magna* and *Ceriodaphnia dubia* were 12 900 and 8900 µg/litre at 20 °C and 17 300 and 11 100 µg/litre at 24 °C, respectively (Cowgill et al., 1985). Rose et al. (1998) reported that *Ceriodaphnia dubia* was approximately 4 times more sensitive to chlorobenzenes than *Daphnia magna*. Richter et al. (1983) compared the acute 48-h LC₅₀ values and chronic 28-day NOEC values of 1,3-DCB and 1,2,4-TCB for *Daphnia magna*. The ratio of acute:chronic toxicity in *Daphnia* was calculated to be 5.1 and 3.0 for 1,3-DCB and 1,2,4-TCB, respectively. Both the number of young produced per adult and the length of adults as expressions of chronic toxicity were equally sensitive for determining statistically significant effects (Richter et al., 1983; De Wolf et al., 1988). The toxicity of chlorobenzenes to midge (*Chironomus riparius*) larvae increased with increasing chlorination. Ratios of LC₅₀:NOEC values were 13, 5.0, 2.3–6.6, and 2.4 for 1,4-DCB, 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB, respectively (Roghair et al., 1994). The toxicity of chlorobenzenes to sand crab (*Portunus pelagicus*) increased during the moulting stage. Toxicity also increased with increasing chlorination and octanol/water partition coefficient (Mortimer & Connell, 1994).

The NOEC, with reference to effects on survival and embryo hatchability for zebrafish (*Brachydanio rerio*) eggs exposed to MCB, 1,4-DCB, 1,2,3-TCB, 1,2,3,4-TeCB, or PeCB, was reported to be the same, regardless of whether the exposure period was 7, 14, 21, or 28 days (Van Leeuwen et al., 1985). As the test solutions were replaced 3 times per week, the identical NOECs are unlikely to be due to evaporation of the chlorobenzenes. The effect of dissolved

oxygen on the toxicity of 1,2,4-TCB to fathead minnows (*Pimephales promelas*) was examined by Carlson (1987). Fish were exposed to 1,2,4-TCB concentrations of up to 920 µg/litre at 4.5, 5.6, or 8.1 mg dissolved oxygen per litre. At all dissolved oxygen concentrations, no effects on survival or growth were reported at 1,2,4-TCB concentrations up to 500 µg/litre. Reduced survival and mean body weight were reported in fish exposed to 920 µg/litre, with toxicity increased in fish from the low dissolved oxygen group. Van Hoogen & Opperhuizen (1988) calculated the chlorobenzene concentration in guppy (*Poecilia reticulata*) tissues following lethality in acute toxicity experiments. Lethal concentrations were the same for 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB, at 2.0–2.5 mmol/kg fish. The toxicity of chlorobenzenes to flounder (*Platichthys flesus*) and sole (*Solea solea*) increased with increasing chlorination and was similar in each species (Furay & Smith, 1995). LC₅₀ values for sheepshead minnow (*Cyprinodon variegatus*) exposed to nominal concentrations of chlorinated benzenes were similar for 48-h, 72-h, and 96-h exposure times. This was especially true for MCB and dichlorobenzenes, suggesting volatilization of the test substance (Heitmuller et al., 1981).

7.2 Terrestrial environment

The toxicity of chlorobenzenes to a terrestrial bacterium modified with the *lux* gene (*Pseudomonas fluorescens*) increased with increasing chlorination. Toxicity also increased with increasing molecular symmetry. EC₅₀ values for inhibited bioluminescence ranged from 0.57 to 118.5 mg/litre for 10 chlorobenzene congeners (Boyd et al., 1998).

EC₅₀ values for growth of lettuce (*Lactuca sativa*) exposed to chlorobenzenes ranged from 2 to 1000 mg/kg soil in a 7-day test and from 1 to >1000 mg/kg soil in a 14-day test. Toxicity increased with increasing chlorination, up to tetrachlorobenzenes. PeCB was less toxic than all congeners, with the exception of MCB (Hulzebos et al., 1993). LC₅₀ values expressed as the test solution ranged from 0.028 to 9.3 mg/litre.

The toxicity of chlorobenzenes to earthworms is presented in Table 5. LC₅₀ values for the earthworms *Eisenia andrei* and *Lumbricus rubellus* range from 75 mg/kg soil for tetrachlorobenzene (isomer not specified) to 1107 mg/kg soil for MCB; however, when the results are expressed as pore water concentrations, the values range from 0.22 µmol/litre for PeCB to 4281 µmol/litre for MCB. Two other species tested appear to be more sensitive, with LC₅₀ values of 0.0016–0.0018 mg/kg soil for 1,2,4-TCB; however, the soil type and pore water concentrations were not given, making a direct comparison with the other earthworm tests difficult.

Table 5: Toxicity of chlorobenzenes to earthworms.

Species	End-point	Medium ^a	Chlorobenzene ^b	Concentration in soil (mg/kg)	Concentration in pore water (µmol/litre)	Reference
<i>Eisenia andrei</i>	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	MCB	240	1453	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	MCB	446	797	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	DCB*	128	121	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	DCB*	229	347	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	TeCB*	75	1.6	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	TeCB*	233	1.2	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	PeCB	134	0.47	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	PeCB	238	0.25	Van Gestel et al. (1991)
<i>Lumbricus rubellus</i>	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	MCB	547	4281	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	MCB	1107	2243	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	DCB*	184	178	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	DCB*	615	1556	Van Gestel et al. (1991)

	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	TeCB*	112	2.3	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	TeCB*	201	1.1	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	PeCB	115	0.43	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	PeCB	201	0.22	Van Gestel et al. (1991)
<i>Eudrilus eugeniae</i>	LC ₅₀	Soil	1,2,4-TCB	0.0016		Callahan et al. (1994)
<i>Allolobaphora tuberculata</i>	LC ₅₀	Soil	1,2,4-TCB	0.0018		Callahan et al. (1994)

^a OM = organic matter.

^b Asterisk (*) indicates isomer not specified.

There is a paucity of data on the toxicity of chlorobenzenes to other terrestrial organisms.

8. EFFECTS EVALUATION

Chlorobenzenes are released to the environment during manufacture or use as an intermediate in the production of other chemicals, as a solvent, or as a degreasing agent. Some chlorobenzenes will be released directly to the environment as a result of their use as carriers for pesticides or as room deodorizers.

Chlorobenzenes released to the environment are likely to be volatilized to the atmosphere, although sorption to soils and sediment may also occur. Chlorobenzenes in the atmosphere will be degraded via photochemical oxidation reactions with hydroxyl radicals. Chlorobenzenes in the aquatic and terrestrial environment will be biodegraded, although they may persist under anaerobic conditions.

Very wide ranges of acute toxicity values have been reported for all types of aquatic organisms exposed to the various chlorobenzene congeners. These are summarized in Figure 1. Care must be taken when interpreting these data, as chlorobenzenes may volatilize from the test system (with the lower chlorinated compounds generally more volatile), thus reducing actual exposure. EC₅₀ values for microorganisms and invertebrates range from 8 to 235 000 µg/litre and from 10 to >530 000 µg/litre, respectively. LC₅₀ values for fish range from 135 µg/litre upwards. The toxicity of chlorobenzenes to aquatic organisms increases with increasing chlorination, increasing by over an order of magnitude over the chlorination range (Figure 2). This is partly due to increased uptake and bioaccumulation of higher chlorinated compounds.

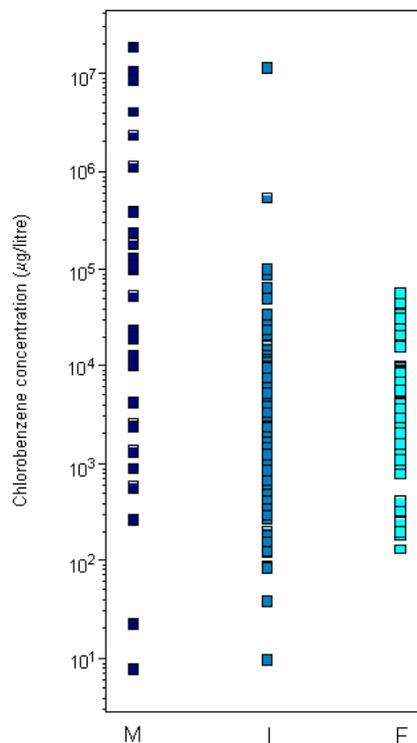


Fig. 1: Acute toxicity of chlorobenzenes (all congeners) to microorganisms (M), invertebrates (I), and fish (F) (data are from Table 3)

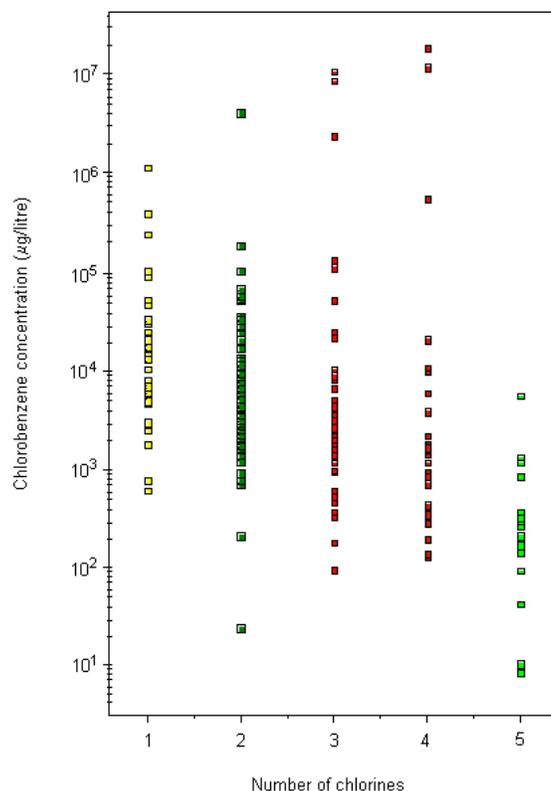


Fig. 2: Acute toxicity of chlorobenzenes to all organisms related to the chlorination of the congeners

Chronic toxicity tests are available for freshwater organisms (algae, invertebrates, and fish). Lowest NOECs, together with the organism and end-point, are presented in Table 6. Long-term studies of marine organisms reporting NOEC values are available for an invertebrate and a fish species.

Table 6: Calculation of risk factors for aquatic organisms.

	MCB	DCBs	TCBs	TeCBs	PeCB
Freshwater					
Lowest freshwater chronic NOEC (µg/litre)	1004 (daphnid reproduction)	550 (daphnid reproduction) ^a	182 (daphnid reproduction)	55 (daphnid reproduction)	32 (daphnid reproduction)
Freshwater PNEC ^b (µg/litre)	20	10	4	1	0.6
Highest measured concentration in fresh water (µg/litre) ^c	>10	>10	8.1	200	0.0006
Risk factor ^d (concentration in fresh water/PNEC)	0.5	1	2	200	0.001
Marine					
Lowest marine chronic NOEC (µg/litre)	125 (crab growth)	31 (crab growth)	25 (crab growth)	17 (crab growth)	5 (crab growth)
Marine/estuarine PNEC ^b (µg/litre)	3	0.6	0.5	0.3	0.1

Highest measured concentration in seawater ($\mu\text{g}/\text{litre}$) ^c	0.5	0.8	5.5	0.02	0.04
Risk factor ^d (concentration in seawater/PNEC)	0.2	1	10	0.07	0.4

^a Fourteen-day EC_{50} value used in the absence of a suitable NOEC.

^b PNEC is calculated by dividing the lowest chronic NOEC by an uncertainty factor of 50 and rounding the calculated value to one significant figure.

^c Concentration values were the highest mean values reported in individual studies. Data from Yamamoto et al. (1997) were not included in this risk assessment, as it was unclear whether the water samples analysed were fresh, estuarine, or marine. Where data were reported as $>10 \mu\text{g}/\text{litre}$, a value of $10 \mu\text{g}/\text{litre}$ was used to calculate the risk factor.

^d Risk factors are rounded to one significant figure.

For compounds studied as extensively as the chlorobenzenes in fresh water, an uncertainty factor of 10 could normally be applied to generate a predicted no-effect concentration (PNEC) for aquatic organisms. However, the lack of chronic NOEC data for algae requires the conservative application of an uncertainty factor of 50. This has been done for both the freshwater and marine data, and results are presented in Table 6.

The concentrations of chlorobenzenes measured in surface waters are presented in Figure 3. This figure includes data points from some additional studies that are not discussed in the text. However, the concentrations in these existing studies fall within the range of concentrations that are discussed in the text. In the 1970s, MCB and dichlorobenzene concentrations in fresh waters ranged up to $10 \mu\text{g}/\text{litre}$, and trichlorobenzene concentrations ranged up to $1 \mu\text{g}/\text{litre}$ (IPCS, 1991a). Chlorobenzene concentrations measured since the late 1980s show a maximum of $0.6 \mu\text{g}/\text{litre}$ for MCB, a maximum of $130 \mu\text{g}/\text{litre}$ for dichlorobenzenes, and a maximum of $10 \mu\text{g}/\text{litre}$ for trichlorobenzenes (Figure 3). Tetrachlorobenzene concentrations in fresh water show a maximum of $200 \mu\text{g}/\text{litre}$, and PeCB, $0.0006 \mu\text{g}/\text{litre}$; these are older measurements (Figure 3).

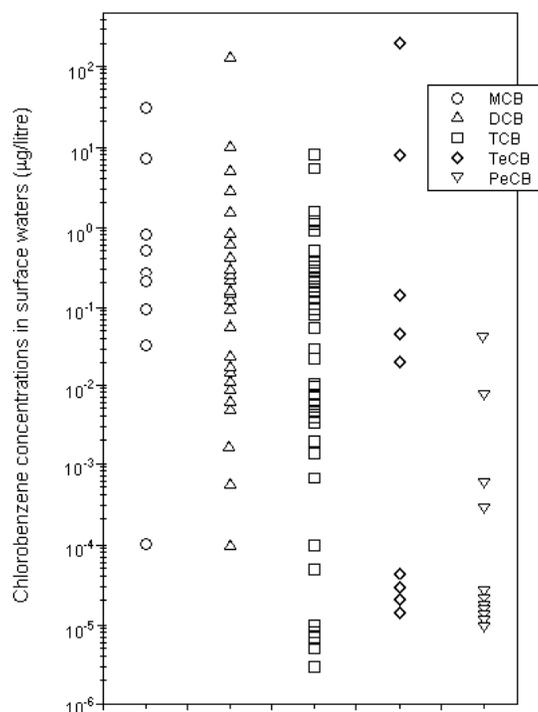


Fig. 3: Concentrations of chlorobenzene congeners in surface waters

Lowest chronic NOECs, PNECs, highest measured water concentrations, and risk factors are summarized in Table 6. Risk factors at, below, or substantially below 1 are generated for all chlorinated benzene congeners with the exception of trichlorobenzenes and tetrachlorobenzenes. Factors of 2 and 10 for trichlorobenzenes in fresh water and seawater, respectively, and a factor of 200 for tetrachlorobenzene in fresh water indicate some risk, particularly for trichlorobenzenes in seawater and tetrachlorobenzenes in fresh water, with the use of the precautionary uncertainty factor of 50.

The data set on freshwater concentrations of chlorobenzenes is limited, and the high risk factors for trichlorobenzenes and tetrachlorobenzenes come from monitoring studies carried out during the early to mid-1980s. The trichlorobenzene concentrations were reported in a single study of Spanish rivers in industrial areas conducted in the mid-1980s. The highest reported concentrations come from one small river with a low flow rate. The highest tetrachlorobenzene concentrations were reported from areas near dump sites. No follow-up monitoring is available for either of these sites. The only conclusion that can be drawn is that under these exceptional circumstances, concentrations of trichlorobenzenes can exceed those likely to produce long-term, but not acute, toxic effects; it can be presumed that these concentrations would arise from point sources. More generally, measured levels of trichlorobenzenes are substantially lower than this, and risk factors would be below 1.

There are more extensive measurements of chlorobenzenes in estuaries and coastal waters. The risk factors in estuarine water exceeding 1 relate to point sources from industrial plants pre-1989. Follow-up monitoring is available for these plants following control of release and/or replacement of chlorobenzenes with alternatives. These more recent data indicate trichlorobenzene concentrations between 1 and 2 orders of magnitude lower. As with fresh water, uncontrolled point source release of trichlorobenzenes will lead to high local risk to organisms.

Terrestrial data, both toxicity studies and measured levels in soil, are inadequate to perform a risk assessment.

9. PREVIOUS EVALUATIONS BY INTERNATIONAL BODIES

Chlorinated benzenes were reviewed by the World Health Organization in 1991 (IPCS, 1991a).

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APPENDIX 1 — SOURCE DOCUMENT

IPCS (1991a) *Chlorobenzenes other than hexachlorobenzene*. Geneva, World Health Organization, International Programme on Chemical Safety (Environmental Health Criteria 128)

A WHO Task Group on Environmental Health Criteria for Chlorobenzenes Other than Hexachlorobenzene met at the Institut d'Hygiène et d'Epidémiologie, Brussels, Belgium, from 25 to 29 June 1990. The Task Group reviewed and revised the draft criteria document and made an evaluation of

the risks for human health and the environment from exposure to chlorobenzenes other than hexachlorobenzene. The drafts of this document were prepared by Ms M.E. Meek and Ms M.J. Giddings, Health Protection Branch, Health and Welfare Canada, Ottawa, Canada. Dr G.C. Becking, IPCS Interregional Research Unit, WHO, Research Triangle Park, NC, USA, was responsible for the overall scientific content of the document, and Mrs M.O. Head, Oxford, England, for the editing. Extensive comments were received from Dr U. Schlottmann, Federal Ministry of the Environment, Germany (chemistry and environmental effects), and Dr R. Fielder, Department of Health, United Kingdom (effects on experimental animals), during the initial review of the document. Dr S. Dobson, Co-Chairman of the Task Group, and Dr P.E.T. Douben made significant contributions and revisions of the draft document during the meeting, particularly the sections dealing with environmental effects.

This CICAD was prepared with reference to the above source document. Additional information from other national assessments was also included:

ATSDR (1990) *Toxicological profile for chlorobenzene*. Atlanta, GA, US Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry.

ATSDR (1998) *Toxicological profile for 1,4-dichlorobenzene (update)*. Atlanta, GA, US Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry.

For more information on these ATSDR assessments, contact:

Agency for Toxic Substances and Disease Registry
Division of Toxicology
1600 Clifton Road NE, Mailstop E-29
Atlanta, GA 30333
USA
Telephone: 1-888-422-8737
Fax: 404-498-0057

APPENDIX 2 — CICAD PEER REVIEW

The draft CICAD on chlorobenzenes other than hexachlorobenzene (environmental aspects) was sent for review to institutions and organizations identified by IPCS after contact with IPCS national Contact Points and Participating Institutions, as well as to identified experts. Comments were received from:

- M. Baril, Institut de Recherche en Santé et en Sécurité du Travail, Montreal, Canada
- R. Benson, Drinking Water Program, US Environmental Protection Agency, Denver, CO, USA
- P. Copestake, Toxicology Advice & Consulting Ltd, Surrey, United Kingdom
- I. Desi, University of Szeged, Szeged, Hungary
- E. Frantik, National Institute of Public Health, Prague, Czech Republic
- A. Juhasz, University of South Australia, Mawson Lakes, Australia
- U. Kierdorf, Justus-Liebig-University of Giessen, Giessen, Germany
- S. Schmidt, Fraunhofer Institute for Toxicology and Experimental Medicine, Hanover, Germany

APPENDIX 3 — CICAD FINAL REVIEW BOARD

Varna, Bulgaria
8–11 September 2003

Members

- Dr I. Benchev, Sofia, Bulgaria
- Dr R. Chhabra, National Institute of Environmental Health Sciences, Research Triangle Park, NC, USA
- Dr C. De Rosa, Agency for Toxic Substances and Disease Registry, Centers for Disease Control and Prevention, Atlanta, GA, USA
- Dr S. Dobson, Centre for Ecology and Hydrology, Monks Wood, Abbots Ripton, Huntingdon, Cambridgeshire, United Kingdom
- Dr G. Dura, National Institute of Environment, József Fodor Public Health Centre, Budapest, Hungary
- Dr L. Fishbein, Fairfax, VA, USA
- Dr H. Gibb, National Center for Environmental Assessment, US Environmental Protection Agency, Washington, DC, USA
- Dr R.F. Hertel, Federal Institute for Risk Assessment, Berlin, Germany
- Mr P. Howe, Centre for Ecology and Hydrology, Monks Wood, Abbots Ripton, Huntingdon, Cambridgeshire, United Kingdom
- Dr S. Ishimitsu, Division of Safety Information on Drug, Food and Chemicals, National Institute of Hygienic Sciences, Tokyo, Japan

Dr D. Kanungo, Central Insecticides Board, Directorate of Plant Protection, Quarantine & Storage, Ministry of Agriculture, Haryana, India

Dr J. Kielhorn, Fraunhofer Institute for Toxicology and Experimental Medicine, Hanover, Germany

Ms B. Meek, Environmental Health Directorate, Health Canada, Ottawa, Ontario, Canada

Dr T. Morita, Division of Safety Information on Drug, Food and Chemicals, National Institute of Hygienic Sciences, Tokyo, Japan

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Dr N. Rizov, National Center of Hygiene, Medical Ecology and Nutrition, Sofia, Bulgaria

Dr P. Schulte, Education and Information Division, National Institute for Occupational Safety and Health, Cincinnati, OH, USA

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Mr P. Watts, Toxicology Advice & Consulting Ltd, Surrey, United Kingdom

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Dr K. Ziegler-Skylakakis, European Commission, Luxembourg

Observers

Dr S. Jacobi, Degussa AG, Fine Chemicals, Hanau-Wolfgang, Germany

Mr M. Southern, Shell International Petroleum Company Ltd, London, United Kingdom

Dr W. ten Berge, DSM, Heerlen, The Netherlands

Secretariat

Dr A. Aitio, International Programme on Chemical Safety, World Health Organization, Geneva, Switzerland

Mr T. Ehara, International Programme on Chemical Safety, World Health Organization, Geneva, Switzerland

APPENDIX 4 — ABBREVIATIONS AND ACRONYMS

ATSDR	Agency for Toxic Substances and Disease Registry (USA)
BCF	bioconcentration factor
CICAD	Concise International Chemical Assessment Document
DCB	dichlorobenzene
EC ₅₀	median effective concentration
ECD	electron capture detector
EHC	Environmental Health Criteria
FID	flame ionization detector
GC	gas chromatography
HDTMA	hexadecyltrimethylamminium
ICSC	International Chemical Safety Card
ILO	International Labour Organization
IPCS	International Programme on Chemical Safety
K _{oa}	octanol/air partition coefficient
	soil sorption coefficient; suspended sediment/water partition coefficient

K_{oc}	
K_{ow}	octanol/water partition coefficient
LC ₅₀	median lethal concentration
LOEC	lowest-observed-effect concentration
MCB	monochlorobenzene
MS	mass spectrometry
NOEC	no-observed-effect concentration
OECD	Organisation for Economic Co-operation and Development
PeCB	pentachlorobenzene
PIM	Poison Information Monograph
PNEC	predicted no-effect concentration
SD	standard deviation
TCB	trichlorobenzene
TeCB	tetrachlorobenzene
TRI	Toxics Release Inventory (USA)
UNEP	United Nations Environment Programme
USA	United States of America
V_{max}	maximum rate of reaction
WHO	World Health Organization

INTERNATIONAL CHEMICAL SAFETY CARDS

[CHLOROENZENE ICSC:0642](#)

[1,2-DICHLOROENZENE ICSC:1066](#)

[1,3-DICHLOROENZENE ICSC:1095](#)

[1,4-DICHLOROENZENE ICSC:0037](#)

[1,2,3-TRICHLOROENZENE ICSC:1222](#)

[1,2,4-TRICHLOROENZENE ICSC:1049](#)

[1,3,5-TRICHLOROENZENE ICSC:0344](#)

[1,2,4,5-TETRACHLOROENZENE ICSC:0676](#)

[PENTACHLOROENZENE ICSC:0531](#)

RÉSUMÉ D'ORIENTATION

Le présent CICAD consacré aux chlorobenzènes autres que l'hexachlorobenzène (aspects environnementaux) est une mise à jour du No 128 de la série Critères d'hygiène de l'environnement (CHE) intitulé *Chlorobenzènes autres que l'hexachlorobenzène* (IPCS, 1991a). Des renseignements sur le devenir et la concentration des chlorobenzènes ont également été tirés de rapports sur le chlorobenzène (ATSDR, 1990) et le 1,4-dichlorobenzène (ATSDR, 1998) publiés par l'Agency for Toxic Substances and Disease Registry. On a poursuivi le dépouillement de la littérature jusqu'en décembre 2002 à la recherche de références complémentaires qui auraient été publiés postérieurement à ces mises au point. L'appendice 1 donne des informations sur la nature de l'examen par des pairs du document initial. Des renseignements sur l'examen par des pairs du présent CICAD sont donnés à l'appendice 2. Ce CICAD a été adopté en tant qu'évaluation internationale lors de la réunion du Comité d'évaluation finale qui s'est tenue à Varna (Bulgarie) du 8 au 11 septembre 2003. La liste des participants à cette réunion figure à l'appendice 3. Les fiches internationales sur la sécurité chimique de divers chlorobenzènes (ICSC 0037, 0344, 0531, 0642, 0676, 1049, 1066, 1095, 1222), établies par le Programme international sur la sécurité chimique (IPCS, 2000, 2003a-h), sont également reproduites dans le présent document. Ce CICAD porte principalement sur les aspects environnementaux de ces composés car depuis la publication du CHE (IPCS, 1991a), l'évaluation des risques pour la santé humaine n'a pas sensiblement changé.

Les dérivés chlorés du benzène ou chlorobenzènes constituent un groupe de composés aromatiques cycliques dans lesquels un atome de chlore est substitué à un ou plusieurs atomes d'hydrogène du noyau benzénique. On les utilise principalement comme intermédiaires dans la synthèse de pesticides ou d'autres produits chimiques. Le 1,4-dichlorobenzène (1,4-DCB) est utilisé comme désodorisant d'ambiance et comme antimites. Les dérivés plus substitués (comme les trichlorobenzènes, le 1,2,3,4-tétrachlorobenzène [1,2,3,4-TeCB] et le pentachlorobenzène [PeCB]) ont été utilisés comme fluides diélectriques.

On n'a pas découvert de sources naturelles de chlorobenzènes. Des chlorobenzènes sont rejetés dans l'environnement lors de leur préparation ou lorsqu'on les utilise comme intermédiaires pour la synthèse d'autres produits. Ils passent également dans l'environnement lorsque des déchets qui en contiennent sont éliminés, par exemple par incinération ou décharge sur des sites mal sécurisés. Le monochlorobenzène (MCB) est rejeté directement dans le milieu ambiant lors de son utilisation comme charge dans certains pesticides. Le passage dans l'environnement des chlorobenzènes utilisés comme désodorisants, fumigants, dégraissants, insecticides, herbicides et défoliants est également la conséquence directe de leur utilisation.

Compte tenu de leurs propriétés physicochimiques, les chlorobenzènes rejetés dans l'environnement se vaporisent vraisemblablement dans l'atmosphère. Ils s'en éliminent ensuite principalement après avoir réagi sur les radicaux hydroxyles pour donner du nitrochlorobenzène, du chlorophénol et des dérivés aliphatiques dicarboxylés, à leur tour éliminés par photolyse ou réaction avec ces mêmes radicaux hydroxyles. Les chlorobenzènes qui passent dans l'environnement aquatique se redistribuent préférentiellement dans l'air et dans les sédiments (notamment les sédiments riches en matières organiques). En solution aqueuse, les chlorobenzènes pourraient théoriquement subir une déchloration réductrice par voie photochimique, mais cette possibilité a été étudiée dans des conditions artificielles qui n'étaient pas représentatives des régions tempérées. Dans le sol, c'est le phénomène de sorption qui constitue le facteur le plus important dont dépendent le devenir et le comportement des chlorobenzènes. Les processus de sorption-désorption qui se déroulent dans le sol conditionnent la vitesse d'évaporation et de lessivage de ces produits, de même que leur faculté de subir une décomposition chimique ou microbienne ou encore d'être fixés par des végétaux ou d'autres organismes.

Dans les divers substrats où ils sont présents - sol, sédiments ou boues d'égout -, les chlorobenzènes sont susceptibles de subir une biodégradation microbienne. Le principal mécanisme de la décomposition anaérobie comporte une déchloration oxydative conduisant à la formation de composés aromatiques hydroxylés (essentiellement des catéchols) suivie de l'ouverture du cycle et d'une minéralisation en dioxyde de carbone et eau. Les chlorobenzènes les moins substitués sont plus facilement dégradés que les autres.

La bioaccumulation de chlorobenzènes par les organismes aquatiques dépend de l'hydro- et de la liposolubilité relative de ces produits (qui correspond au coefficient de partage octanol/eau) ainsi que du degré de substitution. Ces composés sont d'autant mieux captés dans l'eau qu'ils sont plus substitués et que la température est plus élevée.

On a décelé la présence de chlorobenzènes (MCB, dichlorobenzènes et trichlorobenzènes) dans l'air ambiant à une concentration moyenne de 0,1 µg/m³, avec des maxima pouvant atteindre 100 µg/m³ dans des sites dangereux. Dans les eaux de surface, la concentration des chlorobenzènes oscille généralement entre des valeurs de l'ordre du nanogramme ou du microgramme par litre, avec des maxima allant jusqu'à 0,2 mg/litre à proximité des sites industriels. La teneur en chlorobenzènes des eaux usées industrielles est sans doute plus élevée et varie selon les procédés industriels mis en oeuvre. Dans les sols non contaminés, la concentration des différents dichlorobenzènes est généralement inférieure à 0,4 mg/kg et celle des autres chlorobenzènes inférieure à 0,1 mg/kg. Dans les sédiments, les teneurs se situent habituellement dans une gamme de valeurs allant du ng/kg au µg/kg, encore que des valeurs de l'ordre du mg/kg aient été observées dans des échantillons prélevés sur des sites industriels.

D'une façon générale, la toxicité pour les organismes aquatiques augmente avec le degré de chloration du noyau benzénique. Les valeurs de la CE₅₀ à 72 h pour les algues vertes vont de 5280 µg/litre dans le cas du 1,3-DCB à 200 000 µg/litre dans le cas du MCB; de même, la CE₅₀ à 48 h pour les diatomées va de 8 à 235 000 µg/litre. En ce qui concerne les invertébrés d'eau douce, les valeurs de la CE₅₀ à 48 h sont comprises entre 10 µg/litre dans le cas du PeCB et >530 000 µg/litre dans le cas du 1,2,4,5-TeCB. Pour les poissons, les valeurs de la CL₅₀ à 96 h vont de 135 µg/litre dans le cas du PeCB à 21 000 µg/litre dans le cas du 1,2,4-trichlorobenzène (1,2,4-TCB). Les concentrations chroniques sans effet observable (NOEC) sur les invertébrés d'eau douce oscillent entre 32 µg/litre dans le cas du PeCB et 19 000 µg/litre dans le cas du MCB; pour les poissons, les valeurs de la NOEC vont de 18 µg/litre avec le PeCB à 8 500 µg/litre avec le MCB.

On ne possède guère de données concernant les effets des chlorobenzènes sur les organismes terrestres. Pour les plantes en cultures hydroponiques ou en cultures sur sol, on a trouvé des valeurs de la CL₅₀ allant respectivement de 0,028 à 9,3 mg/litre et de 1 à plus de 1000 mg/kg de sol. Pour les lombrics *Eisenia andrei* et *Lumbricus rubellus*, on a mesuré des valeurs de la CL₅₀ qui sont comprises entre 0,22 µmol/litre (eau des pores) dans le cas du PeCB et 4281 µmol/litre dans le cas du MCB.

Il y a peu de chances que les chlorobenzènes aient des effets nocifs sur les organismes aquatiques. Les facteurs de risque obtenus en comparant les valeurs de la toxicité chronique aux concentrations mesurées dans l'environnement sont généralement inférieurs à 1, sauf pour quelques composés pour lesquels ils sont plus élevés, avec la valeur maximum de 200. Les facteurs de risque les plus élevés ont été obtenus en utilisant d'anciennes données portant sur des sources ponctuelles et ne sont donc pas représentatifs de l'environnement dans son ensemble, notamment si l'on tient compte de l'évaporation. On ne disposait pas de données suffisantes pour évaluer le risque auquel sont exposées les espèces terrestres.

RESUMEN DE ORIENTACIÓN

Este CICAD sobre los clorobencenos distintos del hexaclorobenceno (aspectos ecológicos) es una actualización del N° 128 de los Criterios de Salud Ambiental (EHC), *Chlorobenzenes other than hexachlorobenzene* (IPCS, 1991a). También se obtuvo información sobre el destino y las concentraciones de los clorobencenos a partir de los informes de la Agencia para el Registro de Sustancias Tóxicas y Enfermedades sobre el clorobenceno (ATSDR, 1990) y el 1,4-diclorobenceno (ATSDR, 1998). Se realizó asimismo una búsqueda bibliográfica hasta diciembre de 2002 para identificar cualquier información que se hubiera publicado después de la terminación de dichos informes. La información sobre el examen colegiado del documento original se presenta en el apéndice 1. La información sobre el examen colegiado de este CICAD aparece en el apéndice 2. Este CICAD se aprobó como evaluación internacional en una reunión de la Junta de Evaluación Final celebrada en Varna (Bulgaria) del 8 al 11 de septiembre de 2003. La lista de participantes en esta reunión figura en el apéndice 3. También se reproducen en este documento las Fichas internacionales de seguridad química para varios clorobencenos distintos (ICSC 0037, 0344, 0531, 0642, 0676, 1049, 1066, 1095, 1222), preparadas por el Programa Internacional de Seguridad de las Sustancias Químicas (IPCS, 2000, 2003a-h). Este CICAD se concentra en los aspectos ecológicos porque no se han registrado cambios significativos para la salud humana desde la publicación de los Criterios de Salud Ambiental (IPCS, 1991a).

Los bencenos clorados son un grupo de compuestos aromáticos cíclicos en los cuales se han sustituido uno o más átomos de hidrógeno del anillo de benceno por un átomo de cloro. Los clorobencenos se utilizan principalmente como intermediarios en la síntesis de plaguicidas y otros productos químicos. El 1,4-diclorobenceno (1,4-DCB) se utiliza en desodorantes ambientales y como repelente de la polilla. Los bencenos más clorados (trichlorobencenos, 1,2,3,4-tetraclorobenceno [1,2,3,4-TeCB] y pentaclorobenceno [PeCB]) se han utilizado como componentes de fluidos dieléctricos.

No se han identificado fuentes naturales de clorobencenos en el medio ambiente. Los clorobencenos se liberan en el medio ambiente durante su fabricación o su utilización como intermediarios en la producción de otras sustancias químicas. También hay emisiones durante la eliminación de productos de

clorobenceno, por ejemplo en los incineradores y en los vertederos de desechos peligrosos. El monoclorobenceno (MCB) se libera directamente en el medio ambiente debido a su utilización como excipiente de plaguicidas. Los clorobencenos que se utilizan como desodorantes, fumigantes, desengrasantes, insecticidas, herbicidas y defoliantes también se liberan en el medio ambiente como resultado directo de su aplicación.

Sus propiedades fisicoquímicas parecen indicar que es probable que los clorobencenos liberados en el medio ambiente se volatilicen en la atmósfera. La eliminación de la atmósfera se produce fundamentalmente por medio de su reacción con radicales hidroxilo para formar nitroclorobenceno, clorofenol y dicarbonilos alifáticos, que posteriormente se degradan por fotólisis o por reacción con radicales hidroxilo. Los clorobencenos liberados en el medio acuático se redistribuyen preferentemente entre el aire y los sedimentos (sobre todo los que tienen abundante materia orgánica). En teoría, los clorobencenos en soluciones acuosas podrían sufrir una dechloración fotoquímica reductora, aunque sólo se han realizado estudios en condiciones artificiales que no eran representativas de las regiones templadas. El factor más importante que afecta al comportamiento y el destino de los clorobencenos en el suelo es la sorción. Los procesos de adsorción y desorción en el suelo afectan a la tasa de volatilización y lixiviación y a la disponibilidad de sustancias químicas para la degradación microbiana y química o para la absorción por las plantas u otros organismos.

Los microorganismos pueden degradar los clorobencenos en distintos sustratos, por ejemplo el suelo, los sedimentos y los lodos cloacales. El principal mecanismo de degradación aerobia es la dechloración oxidativa, que da lugar a la formación de compuestos aromáticos hidroxilados (sobre todo catecoles), cuyo anillo se rompe y se produce la mineralización posterior hasta dióxido de carbono y agua. Los bencenos menos clorados se degradan más fácilmente que los más clorados.

La bioacumulación de clorobencenos por los organismos acuáticos depende de su solubilidad relativa en el agua y los lípidos (reflejando de esta manera los coeficientes de reparto octanol/agua) y del número de sustituciones de cloro. La absorción a partir del agua aumenta con el grado de cloración y la temperatura.

Se ha notificado la presencia de clorobencenos (MCB, diclorobencenos y triclorobencenos) en el aire exterior, con concentraciones medias del orden de $0,1 \mu\text{g}/\text{m}^3$ y máximas (en vertederos de desechos peligrosos) de hasta $100 \mu\text{g}/\text{m}^3$. Las concentraciones de clorobencenos en las aguas superficiales suelen estar en la escala de ng/l a $\mu\text{g}/\text{l}$, con concentraciones máximas de hasta $0,2 \text{ mg}/\text{l}$ en zonas próximas a fuentes industriales. Las concentraciones de clorobencenos en las aguas residuales industriales pueden ser más elevadas y variar en función del tipo de proceso utilizado. Sus concentraciones en suelos no contaminados son en general inferiores a $0,4 \text{ mg}/\text{kg}$ de diclorobencenos y inferiores a $0,1 \text{ mg}/\text{kg}$ para otros clorobencenos. Las concentraciones de clorobencenos en los alimentos suelen estar en la escala de ng/kg a $\mu\text{g}/\text{kg}$, aunque se ha informado de concentraciones del orden de mg/kg en muestras procedentes de zonas industriales.

En general, la toxicidad en el medio acuático aumenta con el grado de cloración del anillo de benceno. La CE_{50} a las 72 h para las algas verdes oscila entre $5280 \mu\text{g}/\text{l}$ para el 1,3-DCB y $200\,000 \mu\text{g}/\text{l}$ para el MCB; de igual forma, la CE_{50} a las 48 h para las diatomeas varía entre 8 y $235\,000 \mu\text{g}/\text{l}$. En los invertebrados de agua dulce, la CE_{50} a las 48 h oscila entre $10 \mu\text{g}/\text{l}$ para el PeCB y $> 530\,000 \mu\text{g}/\text{l}$ para el 1,2,4,5-TeCB. La CL_{50} a las 96 h para los peces varía entre $135 \mu\text{g}/\text{l}$ para el PeCB y $21\,000 \mu\text{g}/\text{l}$ para el 1,2,4-triclorobenceno (1,2,4-TCB). Las concentraciones sin efectos crónicos observados (NOEC) para los invertebrados de agua dulce oscilan entre $32 \mu\text{g}/\text{l}$ para el PeCB y $19\,000 \mu\text{g}/\text{l}$ para el MCB; en los peces, las NOEC varían entre $18 \mu\text{g}/\text{l}$ para el PeCB y $8500 \mu\text{g}/\text{l}$ para el MCB.

Hay pocos datos disponibles sobre los efectos de los clorobencenos en los sistemas terrestres. Los valores de las CL_{50} para las plantas de cultivos hidropónicos y cultivadas en el suelo oscilan entre $0,028$ y $9,3 \text{ mg}/\text{l}$ y entre 1 y $> 1000 \text{ mg}/\text{kg}$ de suelo, respectivamente. Los valores de las CL_{50} para las lombrices *Eisenia andrei* y *Lumbricus rubellus* varían entre $0,22 \mu\text{moles}/\text{l}$ (agua intersticial) para el PeCB y $4281 \mu\text{moles}/\text{l}$ para el MCB.

El riesgo de que los bencenos clorados provoquen daños en los organismos acuáticos es bajo. Los factores de riesgo comparando los valores de la toxicidad crónica con las concentraciones medidas en el medio ambiente son en general inferiores a 1, salvo algunos compuestos que tenían factores de riesgo más altos, con un valor máximo de 200. Los factores de riesgo más elevados se obtuvieron utilizando antiguos datos de fuentes puntuales, por lo que no son representativos del medio en su conjunto, en particular cuando se tiene en cuenta la probabilidad de evaporación. No se dispuso de datos suficientes para realizar una evaluación del riesgo en las especies terrestres.

ENDNOTES:

1. International Programme on Chemical Safety (1994) *Assessing human health risks of chemicals: derivation of guidance values for health-based exposure limits*. Geneva, World Health Organization (Environmental Health Criteria 170) (also available at <http://www.who.int/pcs/>).

See Also:

[Toxicological Abbreviations](#)



[Canada Gazette](#) > [1998-2007](#) > [Part II: 2006](#) > [2006-11-29](#)

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Registration
SOR/2006-279 November 9, 2006

CANADIAN ENVIRONMENTAL PROTECTION ACT, 1999

Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes)

P.C. 2006-1298 November 9, 2006

Whereas, pursuant to subsection 332(1) ([see footnote a](#)) of the *Canadian Environmental Protection Act, 1999* ([see footnote b](#)), the Minister of the Environment published in the *Canada Gazette*, Part I, on July 9, 2005, a copy of the proposed *Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes)*, substantially in the annexed form, and persons were given an opportunity to file comments with respect to the proposed Regulations or to file a notice of objection requesting that a board of review be established and stating the reasons for the objection;

Whereas, pursuant to subsection 93(3) of that Act, the National Advisory Committee has been given an opportunity to provide its advice under section 6 ([see footnote c](#)) of that Act;

And whereas, in the opinion of the Governor in Council, pursuant to subsection 93(4) of that Act, the proposed Regulations do not regulate an aspect of a substance that is regulated by or under any other Act of Parliament in a manner that provides, in the opinion of the Governor in Council, sufficient protection to the environment and human health;

Therefore, Her Excellency the Governor General in Council, pursuant to subsection 93(1) of the *Canadian Environmental Protection Act, 1999* ([see footnote d](#)), on the recommendation of the Minister of the Environment and the Minister of Health, hereby makes the annexed *Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes)*.

REGULATIONS AMENDING THE PROHIBITION OF CERTAIN TOXIC SUBSTANCES REGULATIONS, 2005 (2-METHOXYETHANOL, PENTACHLOROBENZENE AND TETRACHLOROBENZENES)

AMENDMENTS

1. Part 1 of Schedule 2 to the *Prohibition of Certain Toxic Substances Regulations, 2005* ([see footnote 1](#)) is amended by adding the following after item 1:

	Column 1	Column 2	Column 3

Item	Toxic Substance	Mixture or Product Containing the Toxic Substance	Concentration Limit of the Toxic Substance
2.	2-Methoxyethanol, which has the molecular formula $C_3H_8O_2$	Diethylene glycol methyl ether, which has the molecular formula $C_5H_{12}O_3$	0.5% (w/w)

2. Part 2 of Schedule 2 to the Regulations is amended by adding the following after item 1:

Item	Column 1 Toxic Substances	Column 2 Permitted Uses
2.	2-Methoxyethanol, which has the molecular formula $C_3H_8O_2$	(a) Coatings for aircraft refinishing (b) Semiconductor manufacturing process
3.	Pentachlorobenzene, which has the molecular formula C_6HCl_5	Any use with any chlorobiphenyls described in item 1 of the List of Toxic Substances in Schedule 1 to the <i>Canadian Environmental Protection Act, 1999</i>
4.	Tetrachlorobenzenes, which have the molecular formula $C_6H_2Cl_4$	Any use with any chlorobiphenyls described in item 1 of the List of Toxic Substances in Schedule 1 to the <i>Canadian Environmental Protection Act, 1999</i>

COMING INTO FORCE

3. These Regulations come into force three months after the day on which they are registered.

REGULATORY IMPACT ANALYSIS STATEMENT

(This statement is not part of the Regulations.)

Description

The purpose of the *Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes)* (hereinafter referred to as the Regulations) is to add pentachlorobenzene (QCB) and tetrachlorobenzenes (TeCBs) to the Prohibited Toxic Substances List in Schedule 2, Part 2 of the *Prohibition of Certain Toxic Substances Regulations, 2005* (hereinafter referred to as the Prohibition Regulations), and to add 2-methoxyethanol (2-ME) to the Prohibited Toxic Substances List in Schedule 2, Parts 1 and 2, of the Prohibition Regulations.

On August 9, 2003, the Ministers of the Environment and of Health published their final decision on the assessment of 2-ME in the *Canada Gazette*, and recommended that 2-ME be added to the List of Toxic Substances in Schedule 1 under the *Canadian Environmental Protection Act, 1999* (CEPA 1999). The final version of the assessment report concluded that 2-ME is harmful to human health. However, 2-ME was not considered to have an immediate or long-term harmful effect on the environment or its biological diversity, neither was it found to constitute a danger to the environment on which life depends. On March 9, 2005 an order was published in the *Canada Gazette*, Part II, adding 2-ME to the List of Toxic Substances in Schedule 1 of CEPA 1999.

The final assessment of QCB and TeCBs was published on April 3, 2004, in the *Canada Gazette*, Part I, and, on August 31, 2005, the substances were added to the List of Toxic Substances in Schedule 1 under CEPA 1999. The assessment report concluded that QCB and TeCBs are harmful to the environment or its biological diversity. Therefore, it was recommended that these substances be declared toxic under CEPA 1999. Moreover, because QCB and TeCBs are considered to be toxic under the Act, and are persistent, bioaccumulative and predominantly the result of human activity, they meet the criteria for virtual elimination under the Toxic Substances Management Policy.

Adding QCB and TeCBs to the Prohibited Toxic Substances List in Schedule 2, Part 2 of the Prohibition Regulations will enact a ban on the manufacture, use, sale, offer for sale and import of QCB and TeCBs or any mixture or product containing these substances but allow use exemptions where they are used with chlorobiphenyls. QCB and TeCBs are found in small amounts in chlorobiphenyl liquid, such as liquid used for servicing equipment containing chlorobiphenyls which is regulated under the *Chlorobiphenyls Regulations* and *Storage of PCB Material Regulations*.

Adding 2-ME to the Prohibited Toxic Substances List in Schedule 2, Parts 1 and 2 of the Prohibition Regulations will impose a similar prohibition, while providing flexibility for exempting uses where human exposure is not a concern and technical alternatives are not available. Part 1 establishes a concentration limit for the presence of 2-ME in diethylene glycol methyl ether (DEGME). Part 2 allows the use of 2-ME in coatings for aircraft refinishing and semiconductor manufacturing processes.

The Prohibition Regulations provide a permit system for cases where:

- there is no technically or economically feasible alternative or substitute available to the applicant for the substance;
- the applicant has taken all necessary measures to minimize or eliminate any harmful effect of the toxic substance on the environment and human health; and,
- a plan has been prepared identifying the measures to be taken by the applicant so that the applicant's continued activity will be in compliance with the Prohibition Regulations.

Permits are issued for a period of 12 months and may be renewed twice for the same purpose or use of the substance.

The amendment to the Prohibition Regulations will ensure that the environment and health of Canadians are protected from the potential harmful effects attributed to these toxic substances.

These Regulations will come into force three months after registration by the Clerk of the Privy Council.

Background

2-Methoxyethanol

The purpose of adding 2-ME to Schedule 2, Parts 1 and 2 of the Prohibition Regulations is to protect the health of Canadians by minimizing the potential for human exposure to this substance. Human exposure to 2-ME results from the use of one consumer product by the general population, as well as from potential air releases, leakages and accidental spills in military and industrial uses. Industrial uses include applications as a solvent, chemical intermediate and dispersion agent. The military has used 2-ME as an anti-icing agent for jet fuel and as a component of decontamination agents. Current information indicates that 2-ME is only being used in one consumer product, a cleaning solvent for white-boards.

The use of 2-ME in the consumer product is estimated to be very small, compared to military and industrial uses, but generates the greatest potential for human exposure. Human exposure would

occur mainly through dermal contact with the product containing 2-ME, but also through inhalation of 2-ME evaporated during and after product use. Several countries have already recognized the health concern associated with 2-ME in consumer products. In particular, the European Union has forbidden the sale of products containing 2-ME to the general public since 1994, and France has banned the use of 2-ME in household products since 1997 and in cosmetics since 1998.

The human health risk posed by 2-ME is primarily associated with developmental and reproductive toxicity, including adverse effects on the development of the fetus at levels that are not toxic to the mother (teratogenic effects) observed in experimental animals. Some of the symptoms that have been identified as potential health outcomes of exposure to 2-ME are the occurrence of miscarriages and stillbirths, low-birth-weight babies, reduced fertility, and endocrine disruptions. The risk assessment report concluded that there are no safe thresholds for exposure to 2-ME. Therefore, the environmental objective is to reduce uncontrolled human exposure to 2-ME to the greatest extent possible.

It is expected that the replacement of 2-ME is technically feasible in most applications. Currently available substitutes also belong to the category of glycol ethers, but present a lower risk to human health than 2-ME. In the consumer product, which poses the highest health risk, and in other uses such as anti-icing agent for jet fuel, 2-ME can be directly replaced by substitutes. Overall, substitution is considered to be technically feasible and economically achievable, given the market prices and relative performances of available substitutes.

The Regulations will offer the necessary measures to protect the health of Canadians from exposure to 2-ME in consumer products.

Pentachlorobenzene and Tetrachlorobenzenes

In 2004, QCB and TeCBs were declared toxic under CEPA 1999, based on the conclusion that QCB and TeCBs are entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or on its biological diversity. Because these substances are persistent, bioaccumulative, predominantly anthropogenic and have been declared toxic, releases of QCB and TeCBs are targeted for virtual elimination.

QCB and TeCBs are known to cause both chronic and acute effects on sediment- and soil-dwelling organisms. In general, sediment-dwelling organisms are more sensitive to these chlorobenzenes than soil-dwelling species, based on toxicity studies to date. Additionally, QCB and TeCBs are subject to atmospheric transport from their sources to remote areas.

Total QCB and TeCBs emissions in the Canadian environment are estimated to be 41.8 kilograms (kg) per year and 68.2 kg/year, respectively. QCB and TeCBs are present in products as impurities or are unintentionally produced through waste incineration. The Canada-wide Standards for dioxins and furans, and the regulatory approaches in other Canadian jurisdictions to either prohibit open burning, (including backyard and barrel burning of household waste) or permit it only under pre-approved conditions, will indirectly contribute to the reduction of QCB and TeCBs emissions.

Minor sources of QCB and TeCBs include wood treatment, pesticide use, dielectric fluids, magnesium production, solvent use and long-range transport. Amendments to the existing *Chlorobiphenyls Regulations*, the Wood Preservation Strategic Options Process, and the *Tetrachloroethylene (Use in Dry Cleaning and Reporting Requirements) Regulations* all provide co-benefits by reducing QCB and TeCBs releases from these sources.

Unintentional releases of QCB and TeCBs to water are controlled through the Canadian Council of Ministers of the Environment's interim chronic exposure water quality guideline at 0.0018 milligrams/litre (mg/L) for TeCBs and 0.006 mg/L for QCB. In addition, movement of wastes containing more than 8 parts per million of chlorobenzenes is controlled under the *Export and*

Import of Hazardous Wastes and Hazardous Recyclable Material Regulations and the Interprovincial Movement of Hazardous Waste Regulations.

Alternatives

2-Methoxyethanol

Status Quo

The health risks associated with 2-ME are serious. Furthermore, there is no safe margin of safety for human exposure. It was hence concluded that the status quo could not be allowed to persist and that action to control exposure of the general population to 2-ME would need to be undertaken.

Economic Instruments

Economic instruments, such as emission trading programs and environmental charges, were considered. Emission-trading programs provide a means for seeking cost-effective solutions to reducing exposure, usually below a pre-determined level. However, there is no emission trading possible in a context of eliminating the potential for human exposure to this substance.

Environmental charges present the advantage that they can be aimed at the firms that produce, import, and/or sell 2-ME or products containing 2-ME. However, because charges under CEPA 1999 can only be raised to cover administration costs, there is a high probability that they will not provide enough of an incentive for firms to change their behaviour, therefore resulting in continued human exposure to 2-ME.

Voluntary Measures

Voluntary measures were considered inappropriate for controlling exposure to 2-ME. Potentially adverse health effects are serious and voluntary measures do not ensure that the use of 2-ME will be discontinued, especially in consumer products.

Regulations prohibiting the manufacture, import, offer for sale, sale and use of 2-ME only in consumer products

Prohibiting the use of 2-ME in consumer products would eliminate the most important current route of human exposure to 2-ME. However, health risks might still persist from industrial and military uses. In addition, industry and the Department of National Defence have been receptive to the health concerns associated with 2-ME, and have indicated that they are in the process of moving away from using this substance.

Adding 2-ME to Schedule 2, Parts 1 and 2 of the Prohibition of Certain Toxic Substances Regulations, 2005

Prohibiting the manufacture, import, sale, offer for sale and use of 2-ME presents the most sound and effective way of controlling human exposure. The Regulations will provide a level playing field and ensure that the environmental objective of reducing the potential for human exposure to 2-ME is achieved. In addition, the Regulations will allow the use of 2-ME in specified uses with virtually no risk to human health and where there are no technical alternatives available.

Pentachlorobenzene and Tetrachlorobenzenes

Adding QCB and TeCBs to Schedule 2, Part 2 of the Prohibition of Certain Toxic Substances Regulations, 2005

OCB and TeCBs have been assessed to be toxic under CEPA 1999. Currently, OCB and TeCBs are not manufactured in Canada. It was concluded that prohibiting the manufacture, import, sale, offer for sale and use of OCB and TeCBs, with exemptions for use with chlorobiphenyls, will ensure that the environment and health of Canadians are protected from the potential harmful effects attributed to these toxic substances.

Benefits and Costs

2-Methoxyethanol

Profile and Scenarios

Currently, there is no 2-ME production in Canada. All of 2-ME used in Canada is generally imported from the United States. The quantity of 2-ME imported showed a significant annual variation during the 1990's, fluctuating between 300 and 1,600 tonnes per year. Starting in 2000, imports of 2-ME stabilized around 400-600 tonnes per year and have since remained at that level.

It is estimated that uses of 2-ME are distributed in the following way: 80 percent are military uses in anti-icing agents for jet fuel and decontamination agents; 15 percent are chemical intermediates; 3 percent are dye dispersion agents; 2 percent are industrial processing solvents and analytical reagents (e.g. pharmaceutical processing, electronics manufacturing, electroplating, photographic chemicals, hydraulic and heat transfer fluids); and a very small amount (~0.1 percent) is used in consumer products ([see footnote 2](#)).

Sectors where 2-ME has been used encompass chemicals, furniture manufacturing, rubber manufacturing, pharmaceutical, photographic, and electronics sections. It is estimated that approximately 20 to 36 companies have recently used or currently use 2-ME.

In order to account for uncertainty, two baseline 2-ME demand scenarios were developed using historical data and trends on 2-ME imports, as well as information on firms that are already planning to reformulate away from 2-ME. A high-demand scenario assumed that imports of 2-ME will be 481 tonnes in 2005, and will then decrease to 470 tonnes by 2014 as a result of planned reductions in the use of 2-ME as a dye dispersion agent. After that, 2-ME imports will remain at 470 tonnes from 2014 until 2030. A low-demand scenario assumed that imports will remain constant at 300 tonnes throughout the entire period of the analysis (2005-2030).

The analysis assumed that once the Regulations enter into force, 2-ME would be immediately phased out.

The substitutes considered in this analysis are presented in Table 1, together with the applications where they might be used, their market price, and the expected substitution ratio for 2-ME.

Table 1: 2-ME substitutes, applications where they might be used, market prices, and substitution ratios

Substances	Applications	2004 List Price (\$/kg)	Substitution Ratio for 2-ME
2-ME	- All	3.49	N/A
DEGME [*]	- Fuel additives / decontamination agents	2.78	1:1
	- Chemical intermediates	2.78	1.1
	- Industrial processing solvents / analytical agents	2.78	1.1
PGME ^{**}	- Dye dispersion agent	3.58	1:2

PGME & PGMEA ^{***} (electrical grade)	- Consumer products	3.58	1:1
	- Industrial processing solvents / analytical agents	3.88	1:2
EcoSoft PE	- Dye dispersion agent	2.64	1:1
DEGBE ^{****}	- Consumer Products	2.81	1:1

* DEGME diethylene glycol methyl ether

** PGME propylene glycol monomethyl ether

*** PGMEA propylene glycol monomethyl ether acetate

**** DEGBE diethylene glycol monobutyl ether

Cost-Benefit Analysis Framework

The following are the relevant categories of costs and benefits considered in the analysis:

1. Industry compliance costs. For the purposes of this assessment, we define industry as importers and industrial users of 2-ME. These costs encompassed:

- Reformulation costs
- Input substitution costs

2. Government costs, including:

- Enforcement costs
- Compliance promotion costs
- Input substitution costs to the Department of National Defence

3. Benefits to Canadian Society:

- Reduced health risk

Costs and benefits were assessed over a 25-year timeframe (2005-2030). It was assumed that the Regulations will come into force in 2006 and that 2-ME will be completely substituted in 2007. The costs and benefits assessed are those that directly or indirectly affect Canada or Canadians. All costs and benefits were expressed in 2004 Canadian Dollars.

Wherever possible, economic impacts were reported as present values. The real social discount rate used was 5.5 percent, and a sensitivity range of 3 percent and 10 percent was then applied. The distributional analysis evaluated the allocation of costs and benefits among sectors and regions. The sensitivity analysis considered uncertainty and risk affecting the discount rate, the relative volume and cost of substitutes, the baseline scenario assumptions, and the scale and monetary valuation of expected health benefits.

Costs to the Private Sector

It is expected that industry will respond to the ban on 2-ME by switching to alternative glycol ethers. Substituting 2-ME with other glycol ethers might require reformulation efforts for some applications. In particular, sectors using 2-ME as a chemical intermediate, reagent and solvent might need to reformulate their products. Some sectors, such as office supply manufacturers, will likely replace 2-ME directly with other glycol ethers without reformulating. In addition, the coatings sector using 2-ME as a dye dispersion agent has already started to reformulate to water-based coatings, which do not contain 2-ME.

Reformulation costs were not quantified in this analysis. However, industrial uses of 2-ME are relatively small, hence potential reformulation costs are not expected to be significant from a firm, sector or economy-wide perspective.

Using market prices and substitution ratios based on technical performance, the incremental input costs to the private sector were calculated. Market prices of most substitutes were found to be lower than 2-ME, and many of them were also found to have similar performance ratios (Table 1). Therefore, the Regulations are expected to result in cost savings for most sectors and/or applications. Nevertheless, it is possible that individual sectors or firms might have to use substitutes with a higher price or a lower performance ratio, which will entail input cost increases.

The final results indicate that overall the Regulations will generate cost savings to the Canadian industry. Total cost savings were estimated at C\$0.5 and C\$1.5 million (\$ 2004), for the low- and high-demand scenarios, respectively.

The Regulations are not expected to require any changes in manufacturing equipment, thus no incremental costs associated with capital investment were included. Firms will not have any administrative requirements, such as reporting or monitoring. Product availability and quality are not predicted to be a problem, as there are readily available substitutes that perform as well as 2-ME and sectors where this could be an issue are being exempted (i.e. coatings for aircraft refinishing and semiconductor manufacturing sectors). Any other costs, such as indirect or transitional costs, were determined to be either non-existent or negligible.

Costs to the Government

Costs to the Government encompass compliance promotion and enforcement costs that Environment Canada will incur in implementing the Regulations. They also include the costs to the Department of National Defence (DND) of replacing 2-ME in its current uses in jet-fuel anti-icing and decontamination agents.

DND will have to phase out the uses of 2-ME in jet fuel additives and decontamination agents. Given that alternative specifications for an anti-icing agent for jet fuel are available that do not contain 2-ME, it is expected that DND will not incur reformulation costs associated with this use. Reformulation of decontamination agents will be required, but cost estimates were not available. In addition, because the available substitute (i.e. DEGME) was cheaper in price and had a similar performance to 2-ME, it is estimated that DND will not incur any costs but rather will see cost savings from lower input costs. The present value of cost savings to DND was estimated at C\$2 million and C\$3.2 million (\$2004) for the low- and high-demand scenarios, respectively.

Compliance promotion activities are intended to encourage the regulated community to achieve compliance. Compliance promotion costs would require an annual budget of C\$20,000 (\$2004) during the first year of coming into force of the Regulations. Given the small size of the regulated community and the nature of the Regulations, compliance promotion activities will be minimal. Activities could include mailing out of the Regulations, answering inquiries and developing and distributing promotional materials explaining the Regulations (e.g. fact sheet, Web site material). In years two and three, compliance promotion activities will be at a maintenance level and will be limited to responding and tracking inquiries as well as contributing to the compliance promotion database. This would require an annual budget of C\$1,000 (\$2004). The net present value of

compliance promotion activities was estimated at C\$20,700 (\$2004). However, it should be noted that a higher level of effort for compliance promotion may be required if, following enforcement activities, compliance with the Regulations is found to be low.

Permits-system administration activities will include developing permits, administering permits, verifying information, and answering inquiries. Since the requests for permits are estimated to be low for the first year and onward, the cost is estimated to be negligible.

Annual enforcement costs are estimated at C\$93,548 per year, starting in 2007. The annual cost is broken down as follows: \$28,198 for inspections, \$46,426 for investigations and \$18,924 for measures to deal with alleged violations (including but not limited to environmental protection compliance orders, injunctions and prosecutions). The present value of enforcement costs for 2007 to 2030 was calculated to be C\$578,930 (\$2004).

Total Costs

The Regulations are expected to result in cost savings to both the private and public sector. The total cost savings to Canadian society was calculated by aggregating all cost savings, including private and public sector. The present value estimates of total cost savings were C\$1,930,000 (\$2004) and C\$4,130,000 (\$2004) for the low- and high-demand scenarios, respectively.

Benefits to Canadians

Health Benefits

The ban on 2-ME will bear the largest benefit in terms of lower risks to human health. In particular, this substance was associated in test animals with reproductive and developmental complications, including adverse effects on the development of the fetus at levels that are not toxic to the mother (teratogenic effects). In contrast, the substitutes that will likely replace 2-ME in the consumer product do not present such risks to human health. These substitutes are not associated with teratogenic effects, and do not present the reproductive effects shown by 2-ME. In addition, glycol ethers that will likely be used as 2-ME replacements in industrial and military applications also present a more benign toxicological profile, including lower dermal absorption rates and higher health risk thresholds than 2-ME.

A quantitative analysis of baseline consumer exposure was conducted using the ConsExpo model developed by the National Institute for Public Health and the Environment in the Netherlands. The parameters used in developing the exposure scenarios included consumer product properties, use-setting characteristics, contact exposure information, evaporation and dermal absorption rates, and physiology and anatomy of users and bystanders. The results indicated that, under normal circumstances, users of whiteboard cleaners that contain 2-ME are being exposed to potentially harmful levels.

Based on an annual estimate of 5,035 bottles of consumer product containing 2-ME used, and assuming that each bottle is used by a different person, it was found that 5,035 people would be exposed to potentially harmful levels of 2-ME every year.

Four health end-points were used to determine the health risk to consumer product users, namely miscarriages, low-birth-weight babies, infertility, and endocrine symptoms. Because several of these end-points are associated with pregnancy, the number of pregnant women and adults seeking to have a child were estimated using birth rate statistics for Canada. From the total of 5,035 people using the consumer product, it was estimated that about 22 pregnant women and about 100 adults seeking to have a child would be exposed to unhealthy levels of 2-ME every year. In addition, it was assumed that all remaining people exposed to 2-ME would suffer endocrine symptoms requiring a yearly visit to the doctor.

The analysis indicated that all pregnant women exposed to 2-ME through the use of the consumer product would either suffer a miscarriage or have a low-birth-weight baby (i.e. about 22 cases of either miscarriage or low-birth-weight baby). In addition, the approximately 100 adults seeking to have a child every year are expected to have to deal with fertility problems.

The Regulations will lead to the use of more benign substances, which do not present any of the health risks attributed to 2-ME. Hence, it is estimated that the aforementioned health risks will be minimized or eliminated. The reduction in health risks was converted to monetary values using the cost of illness approach. The only health outcome that was not monetized was the risk of miscarriages, due to the lack of guidance and monetary estimates in the academic literature and the work of other regulatory entities.

The cost of illness approach reflects the costs of treating an illness, incurred by both the individual and the society. Estimates in the Cost of Illness Handbook of the United States Environmental Protection Agency (available at www.epa.gov/oppt/coi) were used to value the reduction in the risk of a low-birth-weight baby. The estimate for reducing the risk of a low-birth-weight baby was C\$150,178 (\$2004). Reducing the risk of a case of infertility was valued at C\$10,330 (\$2004), based on the average cost of a basic in vitro fertilization cycle in Canada. The cost of illness of a case of endocrine symptoms was assumed to be the cost of a doctor's visit, which was estimated to be C\$62 (\$2004). Because these values do not include all possible costs related to the health risk, they should be considered lower bounds of actual benefits.

Total Benefits

The present value of total benefits was estimated at C\$33.4 million (\$2004), with a range of \$16-48 million (\$2004). In addition, it was estimated that up to 22 cases of miscarriages will be avoided by establishing a ban on 2-ME, which were not monetized.

Net Benefits

Net benefits to Canadian society were estimated to be positive. The present value of net benefits was calculated to be in the range of C\$17,9 to C\$52,1 million (\$2004), depending on the demand and health outcome scenario considered, using a discount rate of 5.5 percent. Because the estimated risk reduction of up to 22 cases of miscarriage could not be monetized, due to methodology and data limitations, it is realistic to assume that the estimates of net benefits provided here are lower bounds of the actual net benefits.

In addition to testing the sensitivity of net benefits to possible demand and health risk scenarios, sensitivity analysis of net benefits was conducted on the discount rate and the input cost of substitutes. The objective of this sensitivity analysis was to determine the confidence in the calculated estimate of net benefits and whether or not the Regulations have inherent risks that may significantly impact the value of the net benefit estimate. The results of the sensitivity analysis indicated that the net benefits remain positive within the ranges of discount rates and input cost of substitutes tested. Therefore, it can be concluded that the Regulations are desirable from an economic-efficiency perspective.

Competitiveness

2-ME is not manufactured in Canada and demand is satisfied entirely with imports. The primary substitutes to 2-ME are other glycol ethers. Most of these substitutes would also be imported. As a result, net impacts on importers are expected to be limited. In addition, the volumes of 2-ME used are very small, in comparison to total trade, so there would be no significant impact on Canada's trade balance, even if the required volume or import price of substitutes were different from 2-ME. Finally, no other upstream or downstream impacts on exports or imports are anticipated.

The analysis indicated that many available substitutes are cost-effective alternatives, while others

might represent cost increases. Given the small quantities of 2-ME used, the Regulations are not expected to have negative impacts in the competitiveness of the Canadian economy. However, individual firms might have difficulties finding appropriate substitutes or might find more costly ones. This analysis did not find substantial information indicating that the implementation of the Regulations would result in employment losses or plant closures. In the absence of significant demand feedbacks, such as reductions in the overall demand for products produced using 2-ME, no impacts on employment in other sectors are anticipated.

Pentachlorobenzene and Tetrachlorobenzenes

QCB and TeCBs are not manufactured in Canada, and there is no commercial domestic demand for these substances. There are also no known natural sources of QCB or TeCBs. These substances are present in products as impurities or are unintentionally produced through waste incineration.

Various initiatives indirectly contribute to reductions in QCB and TeCBs emissions, such as:

- the Canada-wide Standards for dioxins and furans;
- the regulatory approaches in other Canadian jurisdictions to either prohibit open burning, or permit it only under pre-approved conditions;
- amendments to the existing *Chlorobiphenyls Regulations*;
- the Wood Preservation Strategic Options Process; and
- the regulations for the control of tetrachloroethylene from the dry-cleaning sector.

The purpose of adding QCB and TeCBs to Schedule 2, Part 2 of the Prohibition Regulations is to ensure the prohibition of the manufacture, use, sale, offer for sale and import of QCB and TeCBs or any mixture or product containing these substances but allow use exemptions where they are used with chlorobiphenyls.

As a consequence, no significant incremental costs or benefits will accrue to the Canadian economy and federal government as a result of placing the two substances on Schedule 2, Part 2 of the Prohibition Regulations.

Consultation

Comments before the pre-publication in the *Canada Gazette*, Part I, on July 9, 2005

2-Methoxyethanol

Public consultation with representatives from environmental non-governmental organizations, industry and other government departments were conducted as part of the risk management process of 2-ME before the proposed Regulations were pre-published in the *Canada Gazette*, Part I.

Overall, stakeholders supported the proposed Regulations. However, one stakeholder expressed concerns about the potential socio-economic impacts of prohibiting the use of 2-ME on the sectors using the substance as a chemical intermediate. The stakeholder also suggested that 2-ME be added to Schedule 2 of the Prohibition Regulations instead of Schedule 1, in order to permit certain uses of 2-ME. This concern has been addressed by the changes to the Regulations.

Pentachlorobenzene and Tetrachlorobenzenes

Public consultation was conducted as part of the risk management process for QCB and TeCBs, before the proposed Regulations were pre-published in the *Canada Gazette*, Part I. The proposed risk management strategy was posted on Environment Canada's Web site for formal consultation with stakeholders of affected sectors (municipal incineration facilities, hazardous waste incineration

facilities, importers and users of perchloroethylene, the Pest Management Regulatory Agency, Industry Canada, Agriculture and Agri-Food Canada, and environmental non-governmental organizations). Comments were received from two facilities and one industry association.

No major issues were raised with regard to the proposed addition of QCB and TeCBs to the *Prohibition of Certain Substances Regulations, 2005*. One facility raised concern over potential economic costs associated with the addition of these substances to the Prohibition Regulations, if they were to apply to products that incidentally contain QCB or TeCBs. The proposed Regulations will not be applicable to products that incidentally contain QCB or TeCBs. It was also commented that the addition of QCB and TeCBs to Schedule 1 of the Prohibition Regulations will not address all of the sources of these two substances.

Comments Following Pre-Publication in the *Canada Gazette*, Part I, on July 9, 2005

2-Methoxyethanol

During the 60-day comment period, a total of five comments were submitted by stakeholders.

Three stakeholders indicated potential impacts in applications where substitutes for 2-ME are not available. These applications included coatings for aircraft refinishing, semiconductor manufacturing processes, and DEGME manufacturing. After reviewing the information provided by the stakeholders concerning the quantities and conditions of use of 2-ME, and the potential releases of 2-ME from industrial processes and manufactured products, Environment Canada and Health Canada concluded that there is virtually no risk to human health associated with the continued use of 2-ME in these applications. Therefore, uses of 2-ME in coatings for aircraft refinishing and semiconductor manufacturing processes were included as permitted uses, and concentration limits were established on the presence of 2-ME in DEGME.

Other commenters were concerned with the analysis of costs and benefits stemming from the Regulations. Environment Canada believes that the analytical framework used and the results achieved are sound and defensible. Hence, no modifications have been made to the analysis of economic impacts.

Pentachlorobenzene and Tetrachlorobenzenes

Following pre-publication of the Regulations in the *Canada Gazette*, Part I, three comments were submitted by stakeholders during the 60-day comment period.

No major issues were raised with regard to the proposed addition of QCB and TeCBs to the *Prohibition of Certain Substances Regulations, 2005*.

It should be noted that QCB and TeCBs have been added to Schedule 2, Part 2 of the Prohibition Regulations instead of Schedule 1. This change is being made to allow the use of QCB and TeCBs in chlorobiphenyl liquid such as liquid used for servicing equipment containing chlorobiphenyls. The current rate of chlorobiphenyl attrition will lead to a discontinuation of the use of the two chlorobenzenes found in equipment containing chlorobiphenyls. Moving QCB and TeCBs to Schedule 2, Part 2 does not impact the original intent of the Regulations. The regulatory text and the Regulatory Impact Analysis Statement have been revised to reflect this change.

Compliance and Enforcement

Since the Regulations are promulgated under CEPA 1999, enforcement officers will, when verifying compliance with the Regulations, apply the Compliance and Enforcement Policy implemented under the Act. The Policy also sets out the range of possible responses to violations, including: warnings, directions, environmental protection compliance orders, ticketing, ministerial orders, injunctions,

prosecution, and environmental protection alternative measures (which are an alternative to a court trial after the laying of charges for a CEPA 1999 violation). In addition, the Policy explains when Environment Canada will resort to civil suits by the Crown for costs recovery.

When, following an inspection or an investigation, an enforcement officer discovers an alleged violation, the officer will choose the appropriate enforcement action based on the following factors:

- *Nature of the alleged violation*: This includes consideration of the damage, the intent of the alleged violator, whether it is a repeat violation, and whether an attempt has been made to conceal information or otherwise subvert the objectives and requirements of the Act.
- *Effectiveness in achieving the desired result with the alleged violator*: The desired result is compliance within the shortest possible time and with no further repetition of the violation. Factors to be considered include the violator's history of compliance with the Act, willingness to co-operate with enforcement officers, and evidence of corrective action already taken.
- *Consistency*: Enforcement officers will consider how similar situations have been handled in determining the measures to be taken to enforce the Act.

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2-Methoxyethanol

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[Footnote a](#)

S.C. 2004, c. 15, s. 31

[Footnote b](#)

S.C. 1999, c. 33

[Footnote c](#)

S.C. 2002, c. 7, s. 124

[Footnote d](#)

S.C. 1999, c. 33

[Footnote 1](#)

SOR/2005-41

[Footnote 2](#)

Information obtained from "2-Butoxyethanol and 2-Methoxyethanol: Current use patterns in Canada, toxicology profiles of alternatives, and feasibility of performing an exposure assessment study", by ToxEcology Environmental Consulting Ltd., May 2003. A copy of the report can be obtained from the contact persons listed at the end of this document

NOTICE:

The format of the electronic version of this issue of the *Canada Gazette* was modified in order to be compatible with extensible hypertext markup language (XHTML 1.0 Strict).

Date Modified: 2010-04-19



CANADA

CONSOLIDATION

CODIFICATION

Prohibition of Certain
Toxic Substances
Regulations, 2005

Règlement sur certaines
substances toxiques
interdites (2005)

SOR/2005-41

DORS/2005-41

Current to November 30, 2010

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OFFICIAL STATUS
OF CONSOLIDATIONS

CARACTÈRE OFFICIEL
DES CODIFICATIONS

Subsections 31(1) and (3) of the *Legislation Revision and Consolidation Act*, in force on June 1, 2009, provide as follows:

Les paragraphes 31(1) et (3) de la *Loi sur la révision et la codification des textes législatifs*, en vigueur le 1^{er} juin 2009, prévoient ce qui suit:

Published
consolidation is
evidence

31. (1) Every copy of a consolidated statute or consolidated regulation published by the Minister under this Act in either print or electronic form is evidence of that statute or regulation and of its contents and every copy purporting to be published by the Minister is deemed to be so published, unless the contrary is shown.

31. (1) Tout exemplaire d'une loi codifiée ou d'un règlement codifié, publié par le ministre en vertu de la présente loi sur support papier ou sur support électronique, fait foi de cette loi ou de ce règlement et de son contenu. Tout exemplaire donné comme publié par le ministre est réputé avoir été ainsi publié, sauf preuve contraire.

Codifications
comme élément
de preuve

...

[...]

Inconsistencies
in regulations

(3) In the event of an inconsistency between a consolidated regulation published by the Minister under this Act and the original regulation or a subsequent amendment as registered by the Clerk of the Privy Council under the *Statutory Instruments Act*, the original regulation or amendment prevails to the extent of the inconsistency.

(3) Les dispositions du règlement d'origine avec ses modifications subséquentes enregistrées par le greffier du Conseil privé en vertu de la *Loi sur les textes réglementaires* l'emportent sur les dispositions incompatibles du règlement codifié publié par le ministre en vertu de la présente loi.

Incompatibilité
— règlements

Registration
SOR/2005-41 February 15, 2005

CANADIAN ENVIRONMENTAL PROTECTION ACT,
1999

**Prohibition of Certain Toxic Substances Regulations,
2005**

P.C. 2005-187 February 15, 2005

Whereas, pursuant to subsection 332(1)^a of the *Canadian Environmental Protection Act, 1999*^b, the Minister of the Environment published in the *Canada Gazette*, Part I, on April 3, 2004, a copy of the proposed *Prohibition of Certain Toxic Substances Regulations, 2005* under the title *Total, Partial or Conditional Prohibition of Certain Toxic Substances Regulations*, substantially in the annexed form, and persons were given an opportunity to file comments with respect to the proposed Regulations or to file a notice of objection requesting that a board of review be established and stating the reasons for the objection;

Whereas, pursuant to subsection 93(3) of that Act, the National Advisory Committee has been given an opportunity to provide its advice under section 6 of that Act;

And whereas, in the opinion of the Governor in Council, pursuant to subsection 93(4) of that Act, the proposed Regulations do not regulate an aspect of a substance that is regulated by or under any other Act of Parliament in a manner that provides, in the opinion of the Governor in Council, sufficient protection to the environment and human health;

Therefore, Her Excellency the Governor General in Council, pursuant to subsection 93(1) of the *Canadian Environmental Protection Act, 1999*^b, on the recommendation of the Minister of the Environment and the Minister of Health, hereby makes the annexed *Prohibition of Certain Toxic Substances Regulations, 2005*.

^a S.C. 2004, c. 15, s. 31

^b S.C. 1999, c. 33

Enregistrement
DORS/2005-41 Le 15 février 2005

LOI CANADIENNE SUR LA PROTECTION DE
L'ENVIRONNEMENT (1999)

**Règlement sur certaines substances toxiques
interdites (2005)**

C.P. 2005-187 Le 15 février 2005

Attendu que, conformément au paragraphe 332(1)^a de la *Loi canadienne sur la protection de l'environnement (1999)*^b, le ministre de l'Environnement a fait publier dans la *Gazette du Canada* Partie I, le 3 avril 2004, sous le titre *Règlement sur l'interdiction totale, partielle ou conditionnelle relative à certaines substances toxiques*, le projet de règlement intitulé *Règlement sur certaines substances toxiques interdites (2005)*, conforme en substance au texte ci-après, et que les intéressés ont ainsi eu la possibilité de présenter leurs observations à cet égard ou un avis d'opposition motivé demandant la constitution d'une commission de révision;

Attendu que, conformément au paragraphe 93(3) de cette loi, le comité consultatif national s'est vu accorder la possibilité de formuler ses conseils dans le cadre de l'article 6 de celle-ci;

Attendu que la gouverneure en conseil est d'avis que, aux termes du paragraphe 93(4) de cette loi, le projet de règlement ne vise pas un point déjà réglementé sous le régime d'une autre loi fédérale de manière à offrir une protection suffisante pour l'environnement et la santé humaine,

À ces causes, sur recommandation du ministre de l'Environnement et du ministre de la Santé et en vertu du paragraphe 93(1) de la *Loi canadienne sur la protection de l'environnement (1999)*^b, Son Excellence la Gouverneure générale en conseil prend le *Règlement sur certaines substances toxiques interdites (2005)*, ci-après.

^a L.C. 2004, ch. 15, art. 31

^b L.C. 1999, ch. 33

PROHIBITION OF CERTAIN TOXIC SUBSTANCES
REGULATIONS, 2005

APPLICATION

1. Subject to sections 2 and 3, these Regulations apply to substances that are both specified on the List of Toxic Substances in Schedule 1 to the *Canadian Environmental Protection Act, 1999* and set out in either Schedule 1 or 2 to these Regulations, referred to in these Regulations as “toxic substances”.

EXCEPTIONS

2. These Regulations do not apply to any toxic substance set out in either Schedule 1 or 2 that is

- (a) contained in a hazardous waste, hazardous recyclable material or non-hazardous waste to which Division 8 of Part 7 of the *Canadian Environmental Protection Act, 1999* applies;
- (b) contained in a control product within the meaning of section 2 of the *Pest Control Products Act*; or
- (c) present as a contaminant in a chemical feedstock used in a process from which there are no releases of the toxic substance and provided that the toxic substance is destroyed or completely converted in that process to a substance that is not a toxic substance set out in either Schedule 1 or 2.

3. (1) These Regulations, except subsections (2), (3) and (4), do not apply to any toxic substance set out in either Schedule 1 or 2 or to any mixture or product containing any such toxic substance that is for use

- (a) in a laboratory for analysis;
- (b) in scientific research; or
- (c) as a laboratory analytical standard.

(2) Any person that intends to use a toxic substance, mixture or product referred to in subsection (1) for a use referred to in that subsection shall, if the quantity of the toxic substance will exceed 10 g in any calendar year,

RÈGLEMENT SUR CERTAINES SUBSTANCES
TOXIQUES INTERDITES (2005)

APPLICATION

1. Sous réserve des articles 2 et 3, le présent règlement s'applique à l'égard des substances mentionnées aux annexes 1 et 2, lesquelles substances sont inscrites sur la liste des substances toxiques de l'annexe 1 de la *Loi canadienne sur la protection de l'environnement (1999)* et sont ci-après désignées «substances toxiques»

EXCEPTIONS

2. Le présent règlement ne s'applique pas à la substance toxique mentionnée aux annexes 1 ou 2 :

- a) qui est contenue dans des déchets dangereux, des matières recyclables dangereuses ou des déchets non dangereux auxquels s'applique la section 8 de la partie 7 de la *Loi canadienne sur la protection de l'environnement (1999)*;
- b) qui est contenue dans un produit antiparasitaire au sens de l'article 2 de la *Loi sur les produits antiparasitaires*;
- c) qui est présente comme contaminant dans une matière première chimique utilisée au cours d'un processus n'occasionnant aucun rejet de la substance toxique, pourvu que celle-ci soit détruite ou totalement convertie au cours de ce processus en une substance autre qu'une substance toxique mentionnée aux annexes 1 ou 2.

3. (1) Le présent règlement, sauf les paragraphes (2) à (4), ne s'applique pas à la substance toxique mentionnée aux annexes 1 ou 2 ni au mélange ou au produit qui en contient, s'ils sont destinés à être utilisés :

- a) pour des analyses en laboratoire;
- b) pour la recherche scientifique;
- c) en tant qu'étalon analytique de laboratoire.

(2) La personne qui prévoit utiliser une substance toxique, un mélange ou un produit visé au paragraphe (1) à l'une des fins visées à ce paragraphe doit, si la quantité de la substance toxique doit dépasser 10 g au

submit to the Minister, at least 30 days before the day on which the person begins using the substance, mixture or product, the information set out in Schedule 3.

(3) Any person that, at the time of coming into force of these Regulations, is using a toxic substance, mixture or product referred to in subsection (1) for a use referred to in that subsection shall, if the quantity of the toxic substance exceeded 10 g in the calendar year immediately preceding the coming into force of these Regulations, submit to the Minister, within 60 days after the date of coming into force of these Regulations, the information set out in Schedule 3.

(4) If, after the coming into force of these Regulations, a toxic substance is added to Schedule 1 or 2, any person that, at the time of the coming into force of the Regulations adding the toxic substance, is using the toxic substance or a mixture or product containing that toxic substance for a use referred to in subsection (1) shall, if the quantity of the toxic substance exceeded 10 g in the calendar year immediately preceding the coming into force of the Regulations adding the toxic substance, submit to the Minister, within 60 days after the date of coming into force of those Regulations, the information set out in Schedule 3.

PROHIBITIONS

4. (1) Subject to section 6, no person shall manufacture, use, sell, offer for sale or import a toxic substance set out in Schedule 1 or a mixture or product containing any such toxic substance unless the substance is incidentally present.

(2) Subsection (1) does not apply to a product that is a manufactured item that is formed into a specific physical shape or design during manufacture and that has, for its final use, a function or functions dependent in whole or in part on its shape or design, if a toxic substance set out in Part 2 of Schedule 1 is present in that manufactured item.

SOR/2010-211, s. 1.

cours d'une année civile, présenter au ministre les renseignements prévus à l'annexe 3, au moins trente jours avant le début de l'utilisation.

(3) La personne qui, à la date d'entrée en vigueur du présent règlement, utilise une substance toxique, un mélange ou un produit visé au paragraphe (1) à l'une des fins visées à ce paragraphe doit, si la quantité de la substance toxique utilisée au cours de l'année civile précédant cette date a dépassé 10 g, présenter au ministre les renseignements prévus à l'annexe 3, dans les soixante jours suivant cette date.

(4) Si une substance toxique est ajoutée aux annexes 1 ou 2 après l'entrée en vigueur du présent règlement, la personne qui, à la date d'entrée en vigueur du règlement visant à ajouter la substance, utilise la substance, ou un mélange ou produit qui en contient, à l'une des fins visées au paragraphe (1) doit, si la quantité de la substance toxique utilisée au cours de l'année civile précédant la date d'entrée en vigueur de ce règlement a dépassé 10 g, présenter au ministre les renseignements prévus à l'annexe 3, dans les soixante jours suivant cette date.

INTERDICTIONS

4. (1) Sous réserve de l'article 6, il est interdit de fabriquer, d'utiliser, de vendre, de mettre en vente ou d'importer une substance toxique mentionnée à l'annexe 1, ou un mélange ou un produit qui en contient, à moins que la substance toxique n'y soit présente fortuitement.

(2) Les produits qui sont des articles manufacturés dotés d'une forme ou de caractéristiques matérielles précises pendant leur fabrication et qui ont, pour leur utilisation finale, une ou plusieurs fonctions en dépendant en tout ou en partie ne sont pas visés par les interdictions prévues au paragraphe (1) à l'égard des substances toxiques mentionnées à la partie 2 de l'annexe 1.

DORS/2010-211, art. 1.

5. Subject to section 6, no person shall manufacture, use, sell, offer for sale or import

(a) a toxic substance set out in column 1 of Part 1 of Schedule 2 if it is present, incidentally or not, in a mixture or product set out in column 2 and if the concentration of the toxic substance in the mixture or product exceeds the limit set out in column 3; or

(b) a toxic substance set out in column 1 of Part 2 of Schedule 2 or a mixture or product containing the toxic substance if the toxic substance, mixture or product is designed for uses other than the uses set out in column 2 unless the substance is incidentally present.

PERMITS

6. (1) Any person that, at the time of coming into force of these Regulations, is manufacturing, using, selling or offering for sale a toxic substance referred to in either section 4 or 5 or a mixture or product containing such a substance or imports such a toxic substance, mixture or product may continue that activity if that person has been issued a permit under subsection (4).

(2) In the case of a toxic substance added to Schedule 1 and referred to in section 4, any person that, at the time of coming into force of the Regulations adding the toxic substance, is carrying out an activity referred to in subsection (1) requires a permit issued under subsection (4) to continue that activity. This rule also applies to any such person in the case of a toxic substance added to Schedule 2 and referred to in section 5.

(3) An application for a permit shall be submitted to the Minister and contain the information set out in Schedule 4.

(4) Subject to subsection (5), the Minister shall issue the permit if the following conditions are met:

(a) there is no technically or economically feasible alternative or substitute available to the applicant, other

5. Sous réserve de l'article 6, il est interdit de fabriquer, d'utiliser, de vendre, de mettre en vente ou d'importer :

a) une substance toxique mentionnée à la colonne 1 de la partie 1 de l'annexe 2, si elle est présente — fortuitement ou non — dans un mélange ou un produit mentionné à la colonne 2 et si sa concentration dans ce mélange ou ce produit est supérieure à celle prévue à la colonne 3;

b) une substance toxique mentionnée à la colonne 1 de la partie 2 de l'annexe 2, ou un mélange ou un produit qui en contient — sauf si sa présence y est fortuite —, si la substance toxique, le mélange ou le produit n'est pas destiné à l'une des utilisations prévues à la colonne 2.

PERMIS

6. (1) La personne qui, à la date d'entrée en vigueur du présent règlement, fabrique, utilise, vend, met en vente ou importe une substance toxique visée aux articles 4 ou 5, ou un mélange ou un produit qui en contient, peut poursuivre son activité si elle y est autorisée en vertu d'un permis délivré aux termes du paragraphe (4).

(2) Dans le cas d'une substance toxique ajoutée à l'annexe 1 et qui est visée à l'article 4, la personne qui, à la date d'entrée en vigueur du règlement visant à ajouter la substance, exerce une activité visée au paragraphe (1) doit, pour poursuivre son activité, obtenir un permis aux termes du paragraphe (4). La même règle s'applique à une telle personne dans le cas d'une substance toxique ajoutée à l'annexe 2 et qui est visée à l'article 5.

(3) La demande de permis est présentée au ministre et comporte les renseignements prévus à l'annexe 4.

(4) Sous réserve du paragraphe (5), le ministre délivre le permis si les conditions suivantes sont réunies :

a) il n'est techniquement ou économiquement pas viable pour le demandeur de substituer à la substance

than a substance regulated under these Regulations, for the toxic substance;

(b) the applicant has taken all necessary measures to minimize or eliminate any harmful effect of the toxic substance on the environment and human health; and

(c) a plan has been prepared respecting the toxic substance, identifying the measures to be taken by the applicant so that the applicant's continued activity will be in compliance with these Regulations, and the period within which the plan is to be fully implemented does not exceed three years from the date on which a permit is first issued to the applicant.

(5) The Minister shall refuse to issue a permit if the Minister has reasonable grounds to believe that the applicant has provided false or misleading information in support of their application.

(6) A permit issued under this section expires 12 months after the day on which it is issued and may, upon application, only be renewed twice for the same purpose or use of the substance.

(7) The Minister shall revoke a permit if the conditions set out in paragraphs (4)(a) to (c) are no longer met or if the Minister has reasonable grounds to believe that the permit holder has provided false or misleading information to the Minister.

(8) The Minister shall not revoke a permit unless the Minister has provided the permit holder with

(a) written reasons for the revocation; and

(b) an opportunity to be heard, by written representation, in respect of the revocation.

REPORTS

7. Every person that manufactures or imports a toxic substance set out in column 1 of Part 3 of Schedule 2 or a mixture or product containing, whether incidentally or not, any such toxic substance shall submit to the Minister the information set out in Schedule 5 within three months after the end of the calendar year during which

toxique une substance qui n'est pas visée par le présent règlement;

b) le demandeur a pris les mesures nécessaires pour éliminer ou atténuer les effets nocifs de la substance toxique sur l'environnement et la santé humaine;

c) un plan a été dressé à l'égard de la substance toxique comportant les mesures que le demandeur s'engage à prendre pour que ses activités soient conformes au présent règlement et le délai prévu pour sa mise à exécution n'excède pas trois ans suivant la date à laquelle le permis est délivré pour la première fois.

(5) Le ministre refuse de délivrer un permis s'il a des motifs raisonnables de croire que le demandeur a fourni des renseignements faux ou trompeurs au soutien de sa demande.

(6) Le permis expire douze mois après la date de sa délivrance et peut, sur demande, être renouvelé au plus deux fois pour le même objet ou pour la même utilisation de la substance toxique.

(7) Le ministre révoque le permis si les conditions prévues aux alinéas (4)a) à c) ne sont plus respectées ou s'il a des motifs raisonnables de croire que le titulaire du permis lui a fourni des renseignements faux ou trompeurs.

(8) Le ministre ne peut révoquer le permis qu'après :

a) avoir avisé par écrit le titulaire des motifs de la révocation;

b) lui avoir donné la possibilité de présenter des observations écrites au sujet de la révocation.

RAPPORTS

7. La personne qui fabrique ou importe une substance toxique mentionnée à la colonne 1 de la partie 3 de l'annexe 2, ou un mélange ou un produit qui en contient — fortuitement ou non —, doit présenter au ministre les renseignements prévus à l'annexe 5 dans les trois mois suivant la fin de l'année civile durant laquelle la sub-

the toxic substance, mixture or product was manufactured or imported if, in that calendar year,

- (a) the quantity of the toxic substance was equal to or greater than that set out in column 2 of that Part, if any;
- (b) the mixture or product contained the toxic substance in an average concentration equal to or greater than that set out in column 3 of that Part, if any; or
- (c) the quantity of the toxic substance and its concentration in the mixture or product were equal to or greater than those set out in column 4 of that Part, if any.

TESTING REQUIREMENTS

8. Any concentration or quantity required to be determined under these Regulations shall be determined, in accordance with generally accepted standards of scientific practice, by a laboratory that is accredited under the International Organization for Standardization standard ISO/IEC 17025: 1999, entitled *General requirements for the competence of testing and calibration laboratories*, as amended from time to time, or by a laboratory that meets an equivalent standard.

CERTIFICATION

9. Any information required to be submitted to the Minister under these Regulations shall be submitted in a form determined by the Minister and accompanied by a certification, dated and signed by the person referred to in those provisions, or the person authorized to act on their behalf, that the information is accurate and complete.

RECORD KEEPING

10. (1) Every person that submits information to the Minister under these Regulations shall keep a copy of that information, the certification and any documents supporting the information, including test data, if appli-

stance toxique, le mélange ou le produit a été fabriqué ou importé si, au cours de cette année :

- a) la quantité de la substance toxique était égale ou supérieure à celle prévue à la colonne 2, le cas échéant;
- b) la moyenne de concentration de la substance toxique dans le mélange ou le produit était égale ou supérieure à celle prévue à la colonne 3, le cas échéant;
- c) la quantité de la substance toxique et sa concentration dans le mélange ou le produit étaient toutes deux égales ou supérieures à celles prévues à la colonne 4, le cas échéant.

DÉTERMINATION DES CONCENTRATIONS ET QUANTITÉS

8. Pour l'application du présent règlement, la concentration et la quantité sont déterminées, conformément aux exigences de pratiques scientifiques généralement reconnues, par un laboratoire qui est accrédité selon la norme de l'Organisation internationale de normalisation ISO/CEI 17025 : 1999, intitulée *Prescriptions générales concernant la compétence des laboratoires d'étalonnage et d'essais*, avec ses modifications successives, ou par un laboratoire qui répond à une norme équivalente.

ATTESTATION

9. Tout renseignement devant être fourni au ministre en application du présent règlement est présenté en la forme fixée par lui et est accompagné d'une attestation, datée et signée par l'intéressé ou par la personne autorisée à agir en son nom, portant que les renseignements sont complets et exacts.

REGISTRES

10. (1) La personne qui présente au ministre des renseignements en application du présent règlement en conserve copie dans un registre, avec l'attestation et les documents à l'appui, y compris les données d'essai, s'il y

cable, for a period of at least five years beginning on the date of the submission of the information.

(2) The information, certification and supporting documents shall be kept at the person's principal place of business in Canada or, on notification to the Minister, at any other place in Canada where the information, certification, documents and test data can be inspected.

REPEAL

11. The *Prohibition of Certain Toxic Substances Regulations, 2003*¹ are repealed.

COMING INTO FORCE

12. These Regulations come into force three months after the day on which they are registered.

a lieu, pendant au moins cinq ans à compter de la date de leur présentation.

(2) Les renseignements, l'attestation et les documents à l'appui sont conservés à l'établissement principal de la personne au Canada ou en tout autre lieu au Canada dont le ministre a été avisé et où ils peuvent être examinés.

ABROGATION

11. Le *Règlement sur certaines substances toxiques interdites (2003)*¹ est abrogé.

ENTRÉE EN VIGUEUR

12. Le présent règlement entre en vigueur trois mois après la date de son enregistrement.

¹ SOR/2003-99

¹ DORS/2003-99

SCHEDULE 1
(Sections 1 to 4 and 6)

PART 1

PROHIBITED TOXIC SUBSTANCES

Item	Toxic Substances
1.	Dodecachloropentacyclo [5.3.0.0 ^{2,6} .0 ^{3,9} .0 ^{4,8}] decane « Mirex »
2.	Polybrominated Biphenyls that have the molecular formula C ₁₂ H _(10-n) Br _n in which «n» is greater than 2
3.	Polychlorinated Terphenyls that have the molecular formula C ₁₈ H _(14-n) Cl _n in which «n» is greater than 2
4.	Bis(chloromethyl) ether that has the molecular formula C ₂ H ₄ Cl ₂ O
5.	Chloromethyl methyl ether that has the molecular formula C ₂ H ₅ ClO
6.	(4-Chlorophenyl) cyclopropylmethanone, O-[(4-nitrophenyl)methyl] oxime that has the molecular formula C ₁₇ H ₁₅ ClN ₂ O ₃
7.	N-Nitrosodimethylamine, which has the molecular formula C ₂ H ₆ N ₂ O
8.	Hexachlorobutadiene, which has the molecular formula C ₄ Cl ₆
9.	Dichlorodiphenyltrichloroethane (DDT), which has the molecular formula C ₁₄ H ₉ Cl ₅

PART 2

PROHIBITED TOXIC SUBSTANCES UNLESS PRESENT IN MANUFACTURED ITEMS

Item	Toxic Substances
1.	Hexane, 1,6-diisocyanato-, homopolymer, reaction products with alpha-fluoro-omega-2-hydroxyethyl-poly(difluoromethylene), C16-20-branched alcohols and 1-octadecanol
2.	2-propenoic acid, 2-methyl-, hexadecyl ester, polymers with 2-hydroxyethyl methacrylate, gamma-omega-perfluoro-C10-16-alkyl acrylate and stearyl methacrylate
3.	2-propenoic acid, 2-methyl-, 2-methylpropyl ester, polymer with butyl 2-propenoate and 2,5 furandione, gamma-omega-perfluoro-C8-14-alkyl esters, tert-Bu benzenecarboxperoxoate-initiated
4.	2-propen-1-ol, reaction products with pentafluoroiodoethane tetrafluoroethylene telomer, dehydroiodinated, reaction products with epichlorohydrin and triethylenetetramine

SOR/2010-211, ss. 2, 3.

ANNEXE 1
(articles 1 à 4 et 6)

PARTIE 1

SUBSTANCES TOXIQUES INTERDITES

Article	Substance toxique
1.	Dodécachloropentacyclo [5.3.0.0 ^{2,6} .0 ^{3,9} .0 ^{4,8}] décane (mirex)
2.	Les biphényles polybromés dont la formule moléculaire est C ₁₂ H _(10-n) Br _n , où «n» est plus grand que 2
3.	Les triphényles polychlorés dont la formule moléculaire est C ₁₈ H _(14-n) Cl _n , où «n» est plus grand que 2
4.	Éther bis(chlorométhylique) dont la formule moléculaire est C ₂ H ₄ Cl ₂ O
5.	Oxyde de chlorométhyle et de méthyle dont la formule moléculaire est C ₂ H ₅ ClO
6.	Le (4-chlorophényle) cyclopropylméthanone, O-[(4-nitrophényle)méthyl]oxime dont la formule moléculaire est C ₁₇ H ₁₅ ClN ₂ O ₃
7.	N-Nitrosodiméthylamine, dont la formule moléculaire est C ₂ H ₆ N ₂ O
8.	Hexachlorobutadiène, dont la formule moléculaire est C ₄ Cl ₆
9.	Dichlorodiphényltrichloroéthane (DDT), dont la formule moléculaire est C ₁₄ H ₉ Cl ₅

PARTIE 2

SUBSTANCES TOXIQUES INTERDITES SAUF SI PRÉSENTES DANS UN ARTICLE MANUFACTURÉ

Article	Substance toxique
1.	1,6-Diisocyanatohexane, homopolymérisé, produits de réaction avec l'alpha fluoro oméga-(2-hydroxyéthyl)-poly(difluorométhylène), des alcools ramifiés en C16-20 et l'octadécan-1-ol
2.	Méthacrylate d'hexadécyle, polymères avec le méthacrylate de 2-hydroxyéthyle, l'acrylate de gamma-oméga-perfluoroalkyle en C10-16 et le méthacrylate de stéaryle
3.	Méthacrylate d'isobutyle, polymérisé avec l'acrylate de butyle, l'anhydride maléique, esters de gamma-oméga-perfluoroalkyle en C8-14, amorcé avec du benzenecarboxperoxoate de tert-butyle
4.	Alcool allylique, produits de réaction avec du pentafluoroiodoéthane et de tétrafluoroéthylène télomérisés, déshydroiodés, produits de réaction avec de l'épichlorhydrine et la triéthylènetétramine

DORS/2010-211, art. 2 et 3.

SCHEDULE 2
(Sections 1 to 3 and 5 to 7)

CONCENTRATION LIMITS, PERMITTED USES AND
REPORTING THRESHOLDS

PART 1

CONCENTRATION LIMITS

Item	Column 1 Toxic Substance	Column 2 Mixture or Product Containing the Toxic Substance	Column 3 Concentration Limit of the Toxic Substance
1.	Hexachlorobenzene	(a) Trichloroethylene	20 ppb
		(b) Tetrachloroethylene	20 ppb
		(c) Tetrachloromethane	20 ppb
		(d) Magnesium salt (by-product from the magnesium industry)	20 ppb
		(e) Magnesium sludge (by-product from the magnesium industry)	20 ppb
		(f) Hydrochloric acid (by-product)	20 ppb
		(g) Ferric chloride	20 ppb
2.	2-Methoxyethanol, which has the molecular formula C ₃ H ₈ O ₂	(h) Ferrous chloride	20 ppb
		Diethylene glycol methyl ether, which has the molecular formula C ₅ H ₁₂ O ₃	0.5% (w/w)

PART 2

PERMITTED USES

Item	Column 1 Toxic Substances	Column 2 Permitted Uses
1.	Benzidine and benzidine dihydrochloride, that have the molecular formula C ₁₂ H ₁₂ N ₂ and C ₁₂ H ₁₂ N ₂ ·2HCl, respectively	(a) Staining for microscopic examination, such as immunoperoxidase staining, histochemical staining or cytochemical staining
		(b) Reagent for detecting blood in biological fluids
		(c) Niacin test to detect some micro-organisms
		(d) Reagent for detecting chloralhydrate in biological fluids
2.	2-Methoxyethanol, which has the molecular formula C ₃ H ₈ O ₂	(a) Coatings for aircraft refinishing
		(b) Semiconductor manufacturing process

ANNEXE 2
(articles 1 à 3 et 5 à 7)

CONCENTRATIONS MAXIMALES, UTILISATIONS PERMISES
ET SEUILS POUR FINS DE RAPPORT

PARTIE 1

CONCENTRATIONS MAXIMALES

Article	Colonne 1 Substance toxique	Colonne 2 Mélange ou produit contenant la substance toxique	Colonne 3 Concentration maximale de la substance toxique
1.	Hexachlorobenzène	a) Trichloroéthylène	20 ppb
		b) tétrachloroéthylène	20 ppb
		c) tétrachlorométhane	20 ppb
		d) sel de magnésium (sous-produit de l'industrie du magnésium)	20 ppb
		e) boue de magnésium (sous-produit de l'industrie du magnésium)	20 ppb
		f) acide chlorhydrique (sous-produit)	20 ppb
		g) chlorure ferrique	20 ppb
2.	2-méthoxyéthanol, dont la formule moléculaire est C ₃ H ₈ O ₂	h) chlorure ferreux	20 ppb
		Éther méthylique de diéthylèneglycol, dont la formule moléculaire est C ₅ H ₁₂ O ₃	0,5 % (p/p)

PARTIE 2

UTILISATIONS PERMISES

Article	Colonne 1 Substance toxique	Colonne 2 Utilisations permises
1.	La benzidine et le dichlorhydrate de benzidine, dont les formules moléculaires sont respectivement C ₁₂ H ₁₂ N ₂ et C ₁₂ H ₁₂ N ₂ ·2HCl	a) Coloration pour l'examen microscopique, telle que la coloration immunoperoxidase, la coloration histochimique et la coloration cytochimique
		b) réactif pour détecter le sang dans les liquides biologiques
		c) test à la niacine pour détecter certains micro-organismes
		d) réactif pour détecter l'hydrate de chloral dans les liquides biologiques
2.	2-méthoxyéthanol, dont la formule moléculaire est C ₃ H ₈ O ₂	a) Revêtement pour la finition d'aéronefs
		b) procédé de fabrication de semi-conducteurs

Item	Column 1 Toxic Substances	Column 2 Permitted Uses
3.	Pentachlorobenzene, which has the molecular formula C_6HCl_5	Any use with any chlorobiphenyls described in item 1 of the List of Toxic Substances in Schedule 1 to the <i>Canadian Environmental Protection Act, 1999</i>
4.	Tetrachlorobenzenes, which have the molecular formula $C_6H_2Cl_4$	Any use with any chlorobiphenyls described in item 1 of the List of Toxic Substances in Schedule 1 to the <i>Canadian Environmental Protection Act, 1999</i>

Article	Colonne 1 Substance toxique	Colonne 2 Utilisations permises
3.	Pentachlorobenzène, dont la formule moléculaire est C_6HCl_5	Utilisation avec un biphenyle chloré visé à l'article 1 de la liste des substances toxiques de l'annexe 1 de la <i>Loi canadienne sur la protection de l'environnement (1999)</i>
4.	Tétrachlorobenzènes, dont la formule moléculaire est $C_6H_2Cl_4$	Utilisation avec un biphenyle chloré visé à l'article 1 de la liste des substances toxiques de l'annexe 1 de la <i>Loi canadienne sur la protection de l'environnement (1999)</i>

PART 3

REPORTING THRESHOLDS

Item	Column 1 Toxic Substance	Column 2 Annual Quantity	Column 3 Annual Average Concentration	Column 4 Annual Quantity and Annual Average Concentration
1.	Hexachlorobenzene			10 g and 10 ppb
2.	Benzidine and benzidine dihydrochloride, that have the molecular formula $C_{12}H_{12}N_2$ and $C_{12}H_{12}N_2 \cdot 2HCl$, respectively	1 kg		

SOR/2006-279, ss. 1, 2.

PARTIE 3

SEUILS POUR FINS DE RAPPORT

Article	Colonne 1 Substance toxique	Colonne 2 Quantité annuelle	Colonne 3 Moyenne de concentration annuelle	Colonne 4 Quantité et moyenne de concentration annuelles
1.	Hexachlorobenzène			10 g et 10 ppb
2.	La benzidine et le dichlorhydrate de benzidine, dont les formules moléculaires sont respectivement $C_{12}H_{12}N_2$ et $C_{12}H_{12}N_2 \cdot 2HCl$	1 kg		

DORS/2006-279, art. 1 et 2.

SCHEDULE 3
(Subsections 3(2) to (4))

INFORMATION RELATED TO THE USE OF CERTAIN TOXIC
SUBSTANCES IN A LABORATORY FOR ANALYSIS, FOR
SCIENTIFIC RESEARCH OR AS A LABORATORY
ANALYTICAL STANDARD

1. Information respecting the laboratory where a toxic substance or a mixture or a product containing it will be or is being used:

(a) the name, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of the laboratory; and

(b) the name, title, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of the person authorized to act on behalf of the laboratory, if any.

2. Information respecting each toxic substance set out in Schedule 1 or 2, and each mixture or product containing the toxic substance:

(a) the name of the toxic substance and the name of the mixture or product containing the toxic substance, if applicable;

(b) the anticipated period of its use;

(c) the estimated quantity to be used in a calendar year and the unit of measurement; and

(d) the identification of each proposed use and each actual use, as the case may be.

ANNEXE 3
(paragraphe 3(2) à (4))

RENSEIGNEMENTS SUR L'UTILISATION DE CERTAINES
SUBSTANCES TOXIQUES À DES FINS D'ANALYSE EN
LABORATOIRE OU DE RECHERCHE SCIENTIFIQUE OU
COMME ÉTALON ANALYTIQUE DE LABORATOIRE

1. Renseignements sur le laboratoire qui utilise ou utilisera la substance toxique ou le mélange ou produit qui en contient :

a) nom, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique du laboratoire;

b) nom, titre, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique de la personne autorisée à agir au nom du laboratoire, s'il y a lieu.

2. Renseignements sur chacune des substances toxiques mentionnées aux annexes 1 et 2 et sur chaque mélange ou produit qui en contient :

a) le nom de la substance toxique et, le cas échéant, le nom du mélange ou du produit qui en contient;

b) la période d'utilisation prévue;

c) la quantité que l'on prévoit utiliser au cours d'une année civile ainsi que l'unité de mesure;

d) une description de chaque utilisation réelle ou projetée, selon le cas.

SCHEDULE 4
(Subsection 6(3))

INFORMATION TO BE CONTAINED IN AN APPLICATION
FOR A PERMIT

1. Information respecting the applicant:

(a) their name, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any; and

(b) the name, title, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of the person authorized to act on behalf of the applicant, if any.

2. In the case of a toxic substance referred to in section 4 of these Regulations or a mixture or product containing any such toxic substance, the following information:

(a) the name of each toxic substance and the name of each mixture or product containing the toxic substance, if applicable;

(b) the estimated quantity to be manufactured, used, sold, offered for sale or imported in a calendar year and the unit of measurement;

(c) the identification of each proposed use, if known; and

(d) if the applicant is a manufacturer, seller or importer, the name, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of each person in Canada to whom the applicant intends to sell a toxic substance or a mixture or product containing such a toxic substance and the identification of each toxic substance, mixture or product.

3. In the case of a toxic substance referred to in section 5 of these Regulations or any mixture or product containing any such toxic substance, the following information:

(a) the name of the toxic substance and the name of the mixture or product containing the toxic substance, if applicable;

(b) in the case of a toxic substance set out in column 1 of Part 1 of Schedule 2, the concentration of the toxic substance in the mixture or product;

(c) the estimated quantity to be manufactured, used, sold, offered for sale or imported in a calendar year and the unit of measurement;

(d) the identification of each proposed use, if known; and

(e) if the applicant is a manufacturer, seller or importer, the name, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of each person in Canada to whom the applicant intends to sell a toxic substance or a mixture or product containing such a toxic substance and the identification of each toxic substance, mixture or product.

4. Evidence that there is no technically or economically feasible alternative or substitute available to the applicant, other than a substance regulated under these Regulations, for the toxic substance.

ANNEXE 4
(paragraphe 6(3))

RENSEIGNEMENTS À FOURNIR DANS LA DEMANDE DE
PERMIS

1. Renseignements sur le demandeur :

a) nom, adresses municipale et postale et numéro de téléphone du demandeur et, le cas échéant, son numéro de télécopieur et son adresse électronique;

b) nom, titre, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique de la personne autorisée à agir au nom du demandeur, s'il y a lieu.

2. S'agissant d'une substance toxique visée à l'article 4 du présent règlement, ou d'un mélange ou produit qui en contient, les renseignements suivants :

a) le nom de la substance toxique et, le cas échéant, le nom du mélange ou du produit qui en contient;

b) la quantité de substance toxique, de mélange ou de produit que le demandeur prévoit fabriquer, utiliser, vendre, mettre en vente ou importer au cours d'une année civile, ainsi que l'unité de mesure;

c) le détail de chaque utilisation projetée, si le demandeur dispose de cette information;

d) si le demandeur est un fabricant, un vendeur ou un importateur, les nom, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique de chaque personne au Canada à qui il projette de vendre la substance toxique, le mélange ou le produit, ainsi que le nom de la substance, du mélange ou du produit en cause.

3. S'agissant d'une substance toxique visée à l'article 5 du présent règlement, ou d'un mélange ou produit qui en contient, les renseignements suivants :

a) le nom de la substance toxique et, le cas échéant, le nom du mélange ou du produit qui en contient;

b) la concentration de la substance toxique dans le mélange ou le produit, s'il s'agit d'une substance mentionnée à la colonne 1 de la partie 1 de l'annexe 2;

c) la quantité de substance toxique, de mélange ou de produit que le demandeur prévoit fabriquer, utiliser, vendre, mettre en vente ou importer au cours d'une année civile, ainsi que l'unité de mesure;

d) le détail de chaque utilisation projetée, si le demandeur dispose de cette information;

e) si le demandeur est un fabricant, un vendeur ou un importateur, les nom, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique de chaque personne au Canada à qui il projette de vendre la substance toxique, le mélange ou le produit, ainsi que le nom de la substance, du mélange ou du produit en cause.

4. Les renseignements qui établissent qu'il ne lui est technique-ment ou économiquement pas viable de substituer à la substance toxique une substance qui n'est pas visée par le présent règlement.

5. Evidence that explains what measures have been taken to minimize or eliminate any harmful effect of the toxic substance on the environment and human health.

6. A description of the plan prepared respecting the toxic substance identifying the measures to be taken so that the applicant's continued activity will be in compliance with these Regulations as well as the period within which the plan is to be implemented.

5. Les renseignements qui établissent que les mesures ont été prises pour éliminer ou atténuer les effets nocifs de la substance toxique sur l'environnement et la santé humaine.

6. Le détail du plan à l'égard de la substance toxique comportant les mesures à prendre pour que les activités du demandeur soient conformes au présent règlement ainsi que le délai prévu pour sa mise à exécution.

SCHEDULE 5
(Section 7)

INFORMATION RELATED TO THE MANUFACTURE OR
IMPORT OF TOXIC SUBSTANCES

1. Information respecting the manufacturer or importer:

(a) their name, civic and postal addresses of principal place of business, e-mail address, if any, telephone number and fax number, if any; and

(b) the name, title, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of the person authorized to act on behalf of the manufacturer or importer, if any.

2. Information respecting each toxic substance set out in column 1 of Part 3 of Schedule 2 and each mixture or product containing the toxic substance manufactured or imported during a calendar year:

(a) the name of the toxic substance and the name of the mixture or the product containing the toxic substance, if applicable;

(b) the calendar year;

(c) the total quantity manufactured, and the unit of measurement;

(d) the total quantity imported, and the unit of measurement;

(e) the quantity sold in Canada, and the unit of measurement;

(f) the identification of each proposed use of the toxic substance and the mixture or product containing the toxic substance, if applicable;

(g) the annual average concentration, if applicable;

(h) the analytical method used to determine the concentration of the toxic substance in the mixture or product, if applicable; and

(i) the analytical method detection limit used to determine the concentration of the toxic substance in the mixture or product, if applicable.

3. The name, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of each person in Canada to whom the manufacturer or importer sold a toxic substance set out in column 1 of Part 3 of Schedule 2 or a mixture or product containing such a toxic substance and the identification of each toxic substance, mixture or product sold.

4. The name, civic and postal addresses, e-mail address, if any, telephone number and fax number, if any, of the laboratory that determined the concentration of the toxic substance in the mixture or product, if applicable.

ANNEXE 5
(article 7)

RENSEIGNEMENTS SUR LA FABRICATION ET
L'IMPORTATION DE SUBSTANCES TOXIQUES

1. Renseignements sur le fabricant ou l'importateur :

a) nom, adresses municipale et postale de l'établissement principal, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique du fabricant ou de l'importateur;

b) nom, titre, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique de la personne autorisée à agir au nom du fabricant ou de l'importateur, s'il y a lieu.

2. Renseignements sur chacune des substances toxiques mentionnées à la colonne 1 de la partie 3 de l'annexe 2 qui est fabriquée ou importée au cours de l'année civile et sur chaque mélange ou produit qui en contient :

a) le nom de la substance toxique et, le cas échéant, le nom du mélange ou produit qui en contient;

b) l'année civile visée;

c) la quantité totale fabriquée, ainsi que l'unité de mesure;

d) la quantité totale importée, ainsi que l'unité de mesure;

e) la quantité vendue au Canada, ainsi que l'unité de mesure;

f) le détail de l'utilisation projetée de la substance toxique et, le cas échéant, du mélange ou produit qui en contient;

g) la moyenne de concentration annuelle, le cas échéant;

h) la méthode analytique utilisée pour déterminer la concentration de la substance toxique dans le mélange ou le produit, le cas échéant;

i) la limite de détection de la méthode analytique utilisée pour déterminer la concentration de la substance toxique dans le mélange ou le produit, le cas échéant.

3. Les nom, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique de chaque personne au Canada à qui le fabricant ou l'importateur a vendu une substance toxique mentionnée à la colonne 1 de la partie 3 de l'annexe 2, ou un mélange ou un produit qui en contenait, ainsi que le nom de la substance, du mélange ou du produit en cause.

4. Les nom, adresses municipale et postale, numéro de téléphone et, le cas échéant, numéro de télécopieur et adresse électronique du laboratoire où la concentration de la substance toxique dans le mélange ou le produit a été déterminée, le cas échéant.

**Response to Request for Information on International Trade
17 December 2010**

Pentachlorobenzene:

Pentachlorobenzene is not produced in Canada and there is no domestic commercial demand for this substance because its use is prohibited.¹

Based on the results of a survey, small amounts of pentachlorobenzene (40 kg during the first 6 months of 1992) were imported into Canada in dielectric fluids for use in the maintenance of transformers.²

¹ Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes). SOR/2006-279. 2006.

² Government of Canada. Environment Canada, Health Canada. Priority Substances List Assessment Report: Pentachlorobenzene. (En40-215/26E). Ottawa, Canada: Canada Communication Group Publishing, 1993



Environment
Canada

Environnement
Canada

RISK MANAGEMENT STRATEGY FOR PENTACHLOROBENZENE (QCB) AND TETRACHLOROBENZENES (TeCBs)

Chemicals Control Branch
Environmental Protection Service
June 2005

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1.0 ISSUE

The first Priority Substances List of the *Canadian Environmental Protection Act* (CEPA) was published in February 1989. This list included Pentachlorobenzene (QCB) and Tetrachlorobenzenes (1,2,4,5-TeCB, 1,2,3,4-TeCB and 1,2,3,5-TeCB). Assessments were performed and completed in 1993. It was concluded that these substances did not present a threat to human life or health or the environment upon which human life depends. However, there was insufficient data to determine whether they did or could pose a threat to the environment.

The *Canadian Environmental Protection Act, 1999* (CEPA 1999), came into effect on March 31, 2000, replacing the original CEPA. Paragraph 64(a) under CEPA 1999 addresses immediate or long-term harmful effects on the environment and also includes effects on biological diversity. CEPA 1999 places more emphasis on pollution prevention and requires special treatment of persistent and bioaccumulative substances that are present in the environment primarily as a result of human activity.

On April 3, 2004, the Ministers of the Environment and Health published the *Follow-up Report on Five PSL1 Substances for Which There Was Insufficient Information to Conclude Whether the Substances Constitute a Danger to the Environment*. QCB and TeCBs were shown to cause both chronic and acute negative effects on benthic and soil-dwelling organisms. The report concludes that both QCB and TeCBs are toxic to the environment under paragraph 64(a) of CEPA 1999.

2.0 BACKGROUND

2.1 Characteristics of Chlorobenzenes

Pentachlorobenzene, CAS No. 608-93-5, and the three isomers of tetrachlorobenzene, 1,2,3,4-tetrachlorobenzene, CAS No. 634-66-2, 1,2,3,5-tetrachlorobenzene, CAS No. 634-90-2 and 1,2,4,5-tetrachlorobenzene, CAS No. 95-94-3, belong to the family of aryl halides that do not exist in nature and are prepared synthetically. As a general rule, the higher the degree of chlorine substitution in the benzene ring, the higher the melting and boiling points of the compound and the more thermally stable the compounds are.

2.2 Production, Import, Use and Sources of Chlorobenzenes

QCB and TeCBs are not currently produced or used in their pure form in Canada, and, currently, there is no domestic commercial demand for these substances. Formerly, they could be found in dielectric fluids used to top up polychlorinated biphenyl (PCB) transformers and in dyestuff carriers. These applications have either been discontinued (dye carriers) or are being phased out (dielectric fluids). A possible source of release may include dielectric PCB material still in use.

The principal current commercial use of QCB is as a chemical intermediate in the formation of pentachloronitrobenzene (also known as quintozone), a fungicide. QCB is present as an impurity in this fungicide. Pentachloronitrobenzene is currently used, but not produced, in Canada. QCB can also be found as an impurity in several herbicides, pesticides and fungicides currently in use in Canada.

Both QCB and TeCBs may be generated when organic compounds are burned or exposed to a large source of energy in the presence of a chlorine source. Through this mechanism, they may be formed and released to the environment as a result of waste incineration and barrel burning of household waste.

3.0 WHY WE NEED ACTION

QCB and all three isomers of TeCB (1,2,4,5-TeCB, 1,2,3,4-TeCB and 1,2,3,5-TeCB) are considered toxic to the environment because they are entering the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity.

The chlorobenzenes under consideration in this strategy are known to cause both chronic and acute negative effects in controlled tests on benthic and soil-dwelling organisms. These chlorobenzenes have also been estimated to persist in sediment for longer than two years. Additionally, QCB and TeCBs may be subject to atmospheric transport from their source to remote areas and, therefore, are considered persistent in air.

Furthermore, these substances meet the criteria of persistence and bioaccumulation, and are present in the environment primarily as a result of human activity. The releases of these substances should be virtually eliminated as they are considered Track 1 substances under the *Toxic Substances Management Policy*.

4.0 EXPOSURE SOURCES

There are no known natural sources of QCB and TeCBs in the environment. There is no commercial activity involving the pure forms of QCB or TeCB in Canada. Below are the known sources of release in decreasing order of significance. An estimate of releases is shown in Table 4.1.

4.1 Major Sources of Release

The sectors responsible for the majority of releases are discussed in this section.

Barrel Burning of Household Waste

A recent report issued by the Burn Barrel Subgroup of the Dioxins/Furans Workgroup of the Great Lakes Bi-national Toxics Strategy has identified that a portion of the municipal solid waste stream is burned on site at residential dwellings in rural areas. The US Environmental Protection Agency (US EPA) has studied emissions from barrel burning. Emission factors applied by A. J. Chandler (2004) show that barrel burning emits far larger amounts of chlorobenzenes, much of which is found in the ash, than incineration and could be a potentially significant source. The estimated releases from barrel burning represent 42% of the total annual releases of QCB and TeCBs.

Dielectric Fluids

As identified by a 2001 Inventory and Technical Study (Cleghorn and Davies), some TeCBs, and trace amounts of QCB, are released to the environment when there are spills of dielectric fluids used for PCB transformers that contain these substances. Such spills would be expected to occur from in-use equipment as opposed to equipment in storage. Dielectric fluids that initially contained PCB products would have small amounts of QCB and TeCBs, while electrical transformers that were topped up with tri- and tetrachlorobenzene blends would contain larger amounts of both substances. It is estimated that 5.6 kg/yr of QCB and 37.5 kg/yr of TeCBs, representing 39% of total annual releases, come from the transformers topped up with these fluids. This estimate was calculated using Environment Canada's 2000 PCB inventory and the Thompson calculation method. It is lower than the estimates in the 1993 assessment reports as a revised calculation method was used and PCB equipment has continued to be taken out of service. When most of the existing PCB equipment is taken out of service (proposed end of use deadlines in the new draft PCB regulations are: December 31, 2009 for equipment containing 500

mg/kg or more PCB and December 31, 2014 for equipment containing between 50 and 500 mg/kg of PCBs and December 31, 2025 for specified equipment containing 50 mg/kg or more of PCBs (Environment Canada, 2005)) the amounts released from spills will tend towards zero.

Pesticides

QCB is used as a chemical intermediate in the production of pentachloronitrobenzene (also known as quintozone). Hexachlorobenzene, has been cited by the US EPA (1997) as an impurity in quintozone, clopyralid, atrazine, chlorothalonil, dacthal, lindane, pentachlorophenols, picloram and simazine. Based on assumptions by the US Environmental Protection Agency (US EPA, 1999), QCB is also found as an impurity in these pesticides. None of these pesticides are produced in Canada. All but dacthal and lindane are currently registered for use in Canada under the *Pest Control Products Act*. Their use may result in the release of QCB into the Canadian environment.

The Pest Management Regulatory Agency (PMRA) has estimated that the release of QCB for 2001 was approximately 6.2 kg, which represents approximately 6% of the total annual releases of chlorobenzenes. This value was calculated based on available sales and production data and the level of QCB contamination in pesticides; as reported by the registrants; and the estimates of QCB contamination in the cases when registrant data were not available. It was assumed that all QCB contamination present in the technical grade active ingredients are transferred into end-use products, that all sales lead to use, and that all uses lead to entry into the environment. This estimate does not include releases of QCB from wood treatment.

Recent efforts of manufacturers to reduce chlorobenzene contamination in pesticides should result in a reduction of these levels.

TeCBs may be found in pesticides as a very negligible contaminant. This source contributes very little or insignificantly to the overall releases of TeCB.

Municipal Solid Waste Incineration

According to the 2001 Inventory and Technical Study (Cleghorn and Davies) waste incineration was identified as a potentially significant source of chlorobenzenes. New data (Chandler, 2004) suggests that controlled municipal solid waste incineration is not a large source of chlorobenzenes releases in Canada. Emissions from controlled municipal solid waste in Canada are regulated by the provincial/territorial governments.

There are some situations (primarily in Newfoundland) where municipal solid waste is incinerated in uncontrolled conditions that would emit far larger amounts of chlorobenzenes than controlled incineration. It is estimated that municipal solid waste incineration contributes approximately 5% of the total annual release of chlorobenzenes.

Hazardous Waste Incineration

QCB and TeCBs have been found in releases from hazardous waste incinerators and cement kilns burning waste-derived fuels. Recent investigation (Chandler, 2004b) indicates that, given the temperatures achieved in the hazardous waste incinerator, there is no scientific or practical basis for suggesting that flow-through emissions of QCB or TeCBs will occur. For the same reasons, any bottom ash residues leaving the incinerator are unlikely to contain QCB or TeCBs since these compounds would volatilize into the gas stream if present in materials charged to the furnace. The only chlorobenzenes released from hazardous waste incineration would therefore come from their creation in the incineration process. It is estimated that releases from this source represent approximately 4% of the total annual release of chlorobenzenes.

Wood Treatment

The 2001 Inventory and Technical Study indicates that QCB is contained in pentachlorophenol as an impurity. Pentachlorophenol is one of the five main wood treatment chemicals that are used in Canada, but it is not manufactured in Canada. In North America, the only currently registered uses for pentachlorophenol are for pressure and thermal treatment of railway ties, utility poles, pilings and outdoor construction materials. Releases from treatment plants include air emissions, which are generally localized, and releases to water and solid wastes in various forms. Treated wood, such as in-service posts and pilings, also releases QCB while in use and once landfilled. It is estimated that 2 kg/yr of QCB are released from wood treatment plants and in service utility poles. This source represents approximately 2% of the total annual releases of chlorobenzenes.

Table 4.1

Estimate of Releases to the Environment of QCB and TeCBs

	Releases (kg/y)			
	Air	Water	Soil	Total
QCB Sources				
Barrel burning of household waste	1.814		20.116	21.93
Wood treatment plants and in service utility poles	2.24		0.1	2.34
Pesticide use			6.2	6.2
Dielectric Fluid spill and cleanup	0.004	4.171	1.451	5.63
Municipal solid waste incineration	0.364		2	2.36
Hazardous waste incineration	1.835			1.84
Magnesium production	1.449	.079		1.53
Solvent use	0.037			0.04
Long-range transport				n/a
Total				41.9
TeCBs Sources	Air	Water	Soil	Total
Dielectric Fluid spill and cleanup	0.023	27.532	9.575	37.13
Barrel burning of household waste	4.265		20.116	24.38
Hazardous waste incineration	3.016			3.02
Municipal solid waste incineration	0.773		2	2.77
Magnesium production	0.364	0.139		0.50
Solvent use	0.037			0.04
Long-range transport				n/a
Total				67.8

4.2 Other Potential Sources of Release

Several other sources or potential sources were identified in the 2001 Inventory and Technical Study (Cleghorn and Davies). In some cases, the available data indicated that these sources contributed very little or insignificantly to the overall releases of QCB and TeCBs. In other cases, data was too scarce to allow a conclusion.

Magnesium Production

The formation of chlorobenzenes is likely to occur during various metals production processes. The magnesium production process includes an electrolytic process which entails carbon and chlorine in contact with each other at high temperature during the electrolysis of magnesium chloride with graphitic electrodes involving the injection of gaseous hydrochloric acid. Releases of QCB and TeCBs to air and water have been reported by the only magnesium plant in Canada using this process. Emission test data from this facility reported the release of 1.53 kg/yr of QCB and 0.50 kg/yr of TeCB. This represents less than 2% of the total annual releases of QCB and TeCBs.

Chlorinated Solvents

The 2001 Inventory and Technical Study (Cleghorn and Davies) indicates that neither QCB nor TeCBs have been detected in the emissions of the only chlorinated solvent manufacturing facility in Canada. However, small amounts of both substances are found in imported perchloroethylene and carbon tetrachloride. The two most important sources of releases for perchloroethylene are dry cleaning and solvent degreasing.

Assuming an average concentration of 5 ppb for both QCB and TeCBs in perchloroethylene, potential releases of QCB and TeCBs due to national use of perchloroethylene would be ~37 g/yr for each substance (Cleghorn and Davies, 2001). Carbon tetrachloride contains a similar amount of chlorobenzenes, but is now only used as a laboratory analytical standard and as feedstock, and, as such, any releases would be negligible.

Secondary Copper and Aluminum Processing

Chlorobenzenes may be formed during secondary copper smelting process (Cleghorn and Davies, 2001). The presence of chlorinated plastics in copper scraps used as a feed to the smelters and the reducing or pyrolytic conditions in blast furnaces are believed to increase the chlorinated dioxin and furan formation. It is likely that QCB and TeCBs could be formed similarly. The secondary aluminum manufacturing process is also suspected to emit QCB and TeCBs to the atmosphere, when remelt aluminum is degassed with hexachloroethane to control magnesium levels. No data were available for these sectors.

Chemical Manufacturing

Hexachlorobenzene (HCB), and thus potentially QCB and TeCBs (U.S. EPA, 1999), releases have been reported (Cleghorn and Davies, 2001) in the manufacture of sodium chlorate and sodium hydroxide by electrolysis when graphite electrodes are used. In these processes, the use of these electrodes has been eliminated in favor of metal electrodes capped with noble metal coating. These changes in the manufacturing process for sodium chlorate and sodium hydroxide have eliminated the potential for chlorobenzenes releases from this source.

Iron and Steel Mills

Iron and steel mill effluents were identified in the 1993 assessment reports as a source of both QCB and TeCBs. Dioxins and furans, as well as octachlorostyrene (OCS) and HCB are formed

and emitted from electric arc furnaces and iron sintering plants, and it is therefore possible that QCB and TeCBs may also be formed. There are very little recent data on the presence of QCB and TeCBs in either the effluents or the stack gases of this sector as these substances are generally not included in the parameters of tests done. One study (Environment Canada and Health Canada, 1993) identified QCB and TeCBs in the stack gases of one mill. The scarcity of data in this sector makes it difficult to conclude as to the importance of its releases.

Petroleum Refineries

Both QCB and TeCBs had been identified in the waste streams of petroleum refineries at the time of the assessment (1993). A study by the U.S. EPA in 1999 found no QCB or TeCBs in U.S. fuels. Changes in the manufacturing process of certain chemicals used in the refining process (ex: caustic soda) have made it unlikely either substance will be generated and emitted to the atmosphere.

Wastewater Treatment Plants

In the 1993 assessment reports, neither QCB nor TeCBs were among the chloryl substances detected in several surveys of municipal wastewater treatment plants in Ontario. QCB and TeCBs have been found in sediments near industrial sources and in the effluent from one wastewater treatment plant. The reason for finding QCB and TeCBs downstream of the wastewater treatment plant is not known. It could be due to their formation in the wastewater treatment plant, but is more likely due to their presence in influent to the wastewater treatment plant. For these reasons, the Inventory and Technical Study concluded that it is not appropriate to calculate discharges of QCB and TeCBs from wastewater treatment plants.

Textile Mills

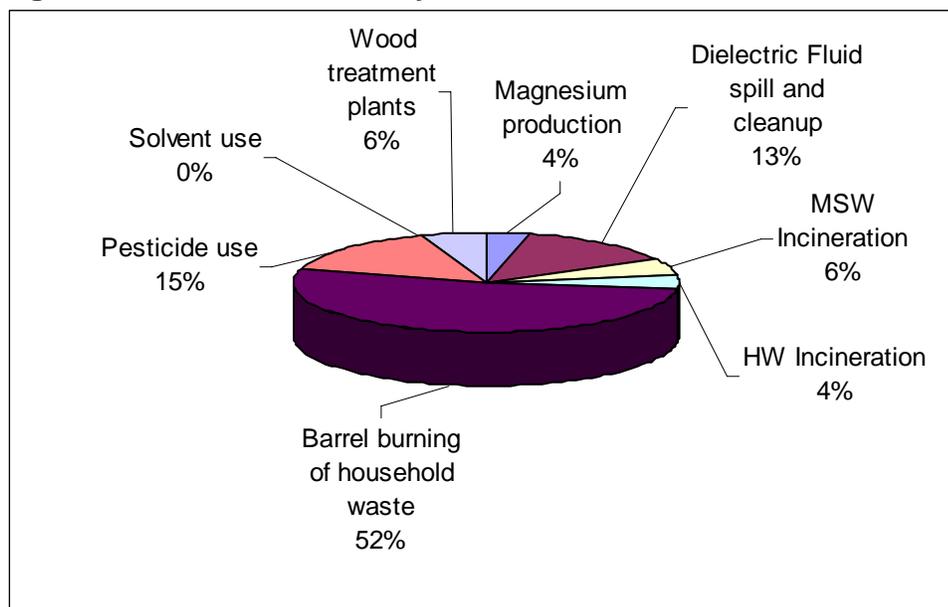
The 1993 assessment reports also found evidence of QCB and TeCBs in the waste stream of a single textile plant. The textile industry uses a large number and a wide range of chemicals, including dye carriers, caustic soda, sodium chlorate and carbon tetrachloride. The use of chlorinated dye carriers has been discontinued and changes in the manufacturing processes for the other chemicals have made it unlikely for them to contain either QCB or TeCBs as impurities. For these reasons, the Inventory and Technical Study (Cleghorn and Davies, 2001) concluded that the data do not support the presence of QCB or TeCBs in effluents from textile mills.

Long-Range Transport

QCB and TeCBs are semi-volatile and persistent substances. Therefore, they can volatilize out of products they are found in, such as pesticides or chlorinated solvents, and undergo long-range transport. For example, QCB from pesticide use outside Canada can enter the Canadian environment by this mechanism. This source is very difficult to quantify, as releases may be transported over some distance, deposit, then re-volatilize to be transported further, leading to double-counting of re-volatilized deposits. It is expected that actual releases from this source would be lower than those estimated in the 1993 assessments as the previous estimates double counted emissions from Canadian sources. Also, extrapolations were used in the previous estimates that are not considered to represent the national situation.

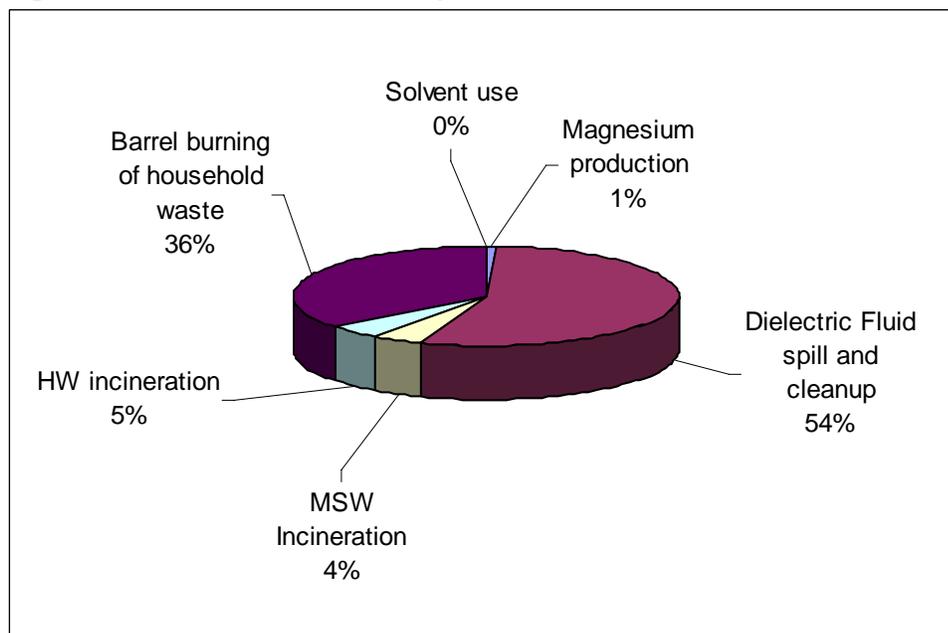
Although the contribution of long-range transport of QCB and TeCBs is not known, it should be noted that there are Canadian and international efforts under way, including Canada-wide Standards for dioxins and furans (Canadian Council of Ministers of the Environment, 2001, 2003, 2003b), virtual elimination of hexachlorobutadiene and other persistent organic pollutants (POPs) and the international POPs Convention, that are expected to reduce emissions of QCB and TeCBs.

Figure 4.1: QCB Releases by Source



Total releases: 41.9 kg/yr

Figure 4.2: TeCB Releases by Source



Total releases: 67.8 kg/yr

5.0 LEGISLATION AND STANDARDS

5.1 Existing Canadian Legislation, Regulations and Guidelines

5.1.1 Releases to Water

The Canadian Council of Ministers of the Environment has established an interim chronic exposure water quality guideline for the protection of freshwater aquatic life of 0.0018 mg/L for 1,2,3,4-TeCB and 0.006 mg/L for QCB.

Ontario regulations developed under the Municipal Industry Strategy for Abatement program require companies in the Organic and Inorganic Chemical Manufacturing Sectors to monitor and report chlorinated benzene levels to the Ministry of the Environment.

5.1.2 Hazardous Waste

The movement of wastes containing 8 ppm or more of chlorobenzenes is controlled under federal *Export and Import of Hazardous Waste Regulations* (1992) and the *Interprovincial Movement of Hazardous Waste Regulations* (2002), and will continue to be controlled under the proposed *Export and Import of Hazardous Waste and Hazardous Recyclable Materials Regulations*.

5.1.3 Guidelines and Standards Targeting Other Chemicals Which May Also Impact Chlorobenzene Emissions

QCB and TeCBs, along with other chlorobenzenes (especially HCB), are often associated with the formation of dioxins and furans. The *Canada-USA Strategy for the Virtual Elimination of Persistent Toxic Substances in the Great Lakes Basin* (known as the Great Lakes Binational Toxics Strategy) requires the United States and Canada to virtually eliminate the Level 1 Substances which are classified as persistent, toxic and bioaccumulative. Dioxins/furans and hexachlorobenzene are classified as Level 1 substances and it is believed that the reduction of releases of these substances can result in the reduction of chlorobenzenes.

The development of Canada-wide Standards for dioxins and furans includes standards for priority sources such as incinerators and municipal waste combustion (Canadian Council of Ministers of the Environment, 2001, 2003). For those dioxins and furans sources that also emit QCB and/or TeCBs, the reduction in dioxins and furans releases afforded by the Canada-wide Standards will likely also reduce the releases of QCB and TeCBs.

Barrel Burning

In several Canadian jurisdictions, a regulatory approach to either prohibit open burning, including backyard and barrel burning of household waste, or permit it only under pre-approved conditions has been adopted. Legislation is used at both the provincial and municipal levels. Nova Scotia's *Solid Waste Resource Management Regulations, 2002* include a ban on open burning of municipal solid waste, including open burning in an uncontrolled teepee, pit or silo burner. Several municipalities and regional districts in British Columbia have passed by-laws that completely prohibit backyard burning or limit burning to dry, garden refuse under strict rules. The government of British Columbia provides municipalities with a model municipal by-law (B.C. Ministry of Water, Land and Air Protection, 1997) to regulate residential backyard burning.

Under the Binational Toxics Strategy for the Great Lakes, a strategy and implementation plan was developed to address the issue of barrel burning. This strategy was developed by the dioxins/furans workgroup, Burn Barrel Subgroup and was finalized in February 2004. While the reduction of dioxin and furan emissions was the driving factor behind the development of this

strategy, the reduction of several other toxic emissions, including chlorobenzenes, is an acknowledged benefit of its implementation. Among the goals of the strategy is the sharing of knowledge and tools with other jurisdictions outside the Great Lakes area to help address this problem nationwide.

PCB Use, Storage and Disposal

After regulations prohibiting new uses of PCB-containing dielectric fluid were introduced in 1980 by Environment Canada, the amount of TeCBs imported for this purpose declined considerably. Revisions to existing regulations on use and storage of PCBs are currently being drafted. Revised requirements on use and storage of PCBs will be consolidated into new proposed PCB Regulations. The proposed PCB Regulations will replace the *Chlorobiphenyls Regulations* and the *Storage of PCB Material Regulations* and, while incorporating most of the original requirements, will add new provisions to end current uses and storage of PCBs and accelerate and track their destruction.

QCB and TeCBs are present in small amounts in the original PCB fluids used, and in larger quantities in the Tri- and TeCB blend used to top up the transformers. As these dielectric fluids are incompatible with new transformers, the gradual elimination of PCB equipment will also lead to their gradual elimination.

Wood Treatment

The substance of concern in the wood treatment industry is pentachlorophenol, which is a registered pesticide. This substance is one of the five pesticides targeted by the Wood Preservation Strategic Options Process. This process led to the development of the *Recommendations for the Design and Operations of Wood Preservation Facilities* (G.E Brudermann) in 1999 and the related facility and process assessment program. One of the objectives is to reduce or eliminate the release of wood preservative chemicals to the environment. A report on the results of the assessment program (Stevens *et al.*, 2001) indicates an average overall conformance level of 68% (36-93% range by criteria) for pentachlorophenol facilities. This conformance level implies a reduction of total chlorobenzenes released to the environment. The *Recommendations for the Design and Operations of Wood Preservation Facilities* were revised and published in 2004. Final conformance assessments are scheduled to be completed by 2005.

Waste Incineration

Canada-wide Standards for the reduction of dioxins and furans emissions from this sector have been developed by the Canadian Council of Ministers of the Environment and were agreed to by the Minister of the Environment (under section 9 of CEPA 1999) in 2001 (for municipal solid waste, hazardous waste, medical waste and sewage sludge incineration) and 2004 (for conical municipal waste combustion). An 86% reduction in releases from municipal solid waste, hazardous waste, medical waste and sewage sludge incineration is expected by 2006. Conical waste combustion will be phased out in Newfoundland by 2008 and new conical waste combustors are prohibited in all jurisdictions in Canada. These efforts should also reduce the amounts of QCB and TeCBs generated by similar mechanisms and emitted from the same sources.

In Ontario, the Ministry of Environment amended Regulation 347 to phase out hospital waste incinerators by December 2003. Hospital wastes are now managed by sending them to a centralized waste incinerator with state-of-the-art technology or by alternative waste management methods.

Perchloroethylene Use

Regulations for the control of perchloroethylene, also deemed a toxic substance under CEPA 1999, were developed by Environment Canada and published in March 2003 for the dry cleaning sector and, in August 2003 for the solvent degreasing sector. By reducing the perchloroethylene releases from these sectors, the QCB and TeCBs releases will also be reduced.

Iron and Steel Mills

The Canada-wide Standards for dioxins and furans were developed to set a goal for reduction of dioxins and furans emissions from electric arc furnaces found in steel manufacturing and from iron sintering plants. They were developed by the Canadian Council of Ministers of the Environment and were agreed to by the Minister of the Environment (under section 9 of CEPA 1999) in 2003. Implementation of the Canada-wide Standards for dioxin and furans emissions should also likely reduce the amounts of QCB and TeCBs generated by similar mechanisms and emitted from the same sources.

5.2 Existing Legislation, Regulations and Guidelines in the United States

5.2.1 Releases as Solid Waste

QCB and 1,2,4,5-TeCB are listed as hazardous constituents under the U.S. *Resource Conservation and Recovery Act*, which requires anyone who generates, transports, treats, stores or disposes of wastes containing these substances to notify the U.S. Environmental Protection Agency (U.S. EPA) within 90 days. These two substances are also part of a list of persistent, bioaccumulative and toxic chemicals which was created to promote voluntary waste minimization efforts under the *Resource Conservation and Recovery Act*.

5.2.2 Releases to Water

Both QCB and 1,2,4,5-TeCB are listed as toxic under the U.S. *Clean Water Act*, and are classified as non-priority substances. National recommended water quality criteria exist for consumption of water and for consumption of aquatic organisms.

QCB, 1,2,4,5-TeCB and 1,2,3,4-TeCB are among the 22 bioaccumulative chemicals of concern identified for further regulation in the U.S. EPA's *Final Rule on the Water Quality Guidance for the Great Lakes System* (2000).

5.2.3 Toxics Control

Under the U.S. *Comprehensive Environmental Response, Compensation and Liability Act*, QCB and 1,2,4,5-TeCB are listed as toxic substances and National Response Center notification is necessary when either substance is released in quantities equal to or exceeding the Reportable Quantity of 2 270 kg for 1,2,4,5-TeCB and 4.54 kg for QCB.

Under the U.S. *Toxic Substances Control Act*, QCB and all three isomers of TeCB are covered by rules which require chemical manufacturers and importers to submit preliminary assessments that include general production, use and exposure information, as well as copies and lists of unpublished health and safety studies on each substance. Also under the *Toxic Substances Control Act*, the testing of 1,2,4,5-TeCB is required to develop health or environmental data.

Under the U.S. *Occupational Health and Safety Act*, 1,2,4,5-TeCB is included on the Right to Know List of 3 States: New Jersey, Pennsylvania and Massachusetts.

5.2.4 Transportation of Dangerous Goods

QCB and 1,2,4,5-TeCB are classified as hazardous materials under the U.S. *Hazardous Materials Transportation Act*, and as such they must be properly labeled and packaged for transportation. A spill notification procedure is also included.

5.3 Existing Legislation, Regulations and Guidelines in Europe

5.3.1 Releases to Water

Under the regulations of the Department of Environment and Transport and the Regions (DETR), the U.K., Scotland and Northern Ireland, it is prohibited to discharge QCB and 1,2,4,5-TeCB directly or indirectly to groundwater.

Under the European Community Water Resource Framework Directive, 1,2,4,5-TeCB is on the list of Dangerous Substances and its discharges into surface and groundwater are to be controlled. No control limits were reported.

5.3.2 Toxics Control

No specific regulatory controls were found for QCB and the three TeCB isomers in the U.K. and the European Union. However, in the European Union, the sale and use of plant protection products containing quintozene with more than 10g/kg of QCB is prohibited.

6.0 PROPOSED OBJECTIVES

6.1 Proposed Environmental Objective

The ultimate environmental objective for tetrachlorobenzenes and pentachlorobenzene is virtual elimination from the environment, as both substances meet the Track 1 criteria set in the *Toxic Substances Management Policy*.

6.2 Proposed Risk Management Objectives

The proposed short-term risk management objective is to prevent the reintroduction of penta- and tetrachlorobenzenes to the Canadian market.

The proposed mid-term risk management objective is to achieve the lowest level of releases technically and economically feasible from the most significant sources of chlorobenzenes.

7.0 PROPOSED RISK MANAGEMENT INSTRUMENT/TOOLS TO BE DEVELOPED

7.1 QCB/TeCBs and Products Containing QCB/TeCBs

Since pentachlorobenzene and tetrachlorobenzenes are persistent, bioaccumulative and toxic substances their reintroduction into the Canadian market must be prevented. In this case, a regulation is an effective way to ensure this objective is met. It is therefore proposed that QCB and TeCBs be added to the *Prohibition of Certain Toxic Substances Regulations, 2005*.

QCB and TeCBs are found as contaminants in certain chlorinated chemicals, therefore, concentration limits should be established, taking into consideration environmental risks and economic and technology factors. Consideration will therefore be given to the development of guidelines to complement the regulations, where environmentally acceptable contamination

levels of chlorobenzenes in products or mixtures will be recommended. Two regulations that came into effect in 2003 (Dry cleaning and Degreasing) will work towards reducing the use of perchloroethylene and its emission to the atmosphere. As a contaminant in perchloroethylene, emissions of chlorobenzenes will also likely be reduced. No additional action is proposed for this sector.

In addition, because QCB and TeCBs meet the criteria of persistence and bioaccumulation and are present in the environment primarily as a result of human activity, Environment Canada will consider adding QCB and TeCB to the Virtual Elimination List along with their LOQ.

QCB and TeCBs are not on the National Pollutant Release Inventory. However, Environment Canada is considering adding them to the inventory in order to monitor progress towards the proposed objectives.

As chlorobenzenes are PSL1 substances, they are not subject to the time limits set out in CEPA 1999 with respect to publication of the proposed and final instrument. However, it is the intention to meet the 2 year objective for the publication of the proposed instrument after the publication of the assessment results, and then the 18 month objective for the subsequent publication of the final instrument.

7.2 QCB and TeCB Releases

Barrel Burning

Barrel burning of household waste is a complex problem to address as jurisdiction over this issue is mainly municipal. It has been recognized as an important source of dioxins and furans, and efforts to address the problem are being made by the provinces of British Columbia and Nova Scotia and through the Great Lakes Binational Toxics Strategy. As dioxins and furans are a human health issue, efforts to control these emissions will likely take a high priority. Any efforts to reduce dioxin and furan emissions from this source will also reduce chlorobenzene emissions.

Due to the various strategies in place for this sector, no additional action is recommended. It is recognized that providing residential waste management infrastructure as an alternative to open burning of garbage in rural areas remains a challenge.

Dielectric Fluids

Two existing regulations related to PCBs (*Storage of PCB Material Regulations*, 1992, and *Chlorobiphenyls Regulations*, 1991) are being revised to address the gradual elimination of PCB materials in use and in storage, and therefore the gradual elimination of other substances present as contaminants in them. No additional action is proposed for this sector.

Wood Treatment

Recommendations for the Design and Operations of Wood Preservation Facilities (Brudermann, 2004) are already in place. All but three facilities have complied with this voluntary program. A proposed pollution prevention planning Notice has been developed to address the three facilities that have not complied. No additional action is proposed for this sector.

Pesticides

Pesticides are regulated under the authority of the *Pest Control Products Act*, which is administered by the Pest Management Regulatory Agency (PMRA). The PMRA will manage QCB contamination in pest control products according to its Regulatory Directive *The Pest*

Management Regulatory Agency's Strategy for Implementing the Toxic Substances Management Policy (Dir99-03).

Under this Directive, levels of micro-contaminants of concern in pest control products are reviewed to ensure that products, and the uses for which they are registered, do not pose unacceptable risks to people or the environment. The PMRA works in partnership with registrants to reduce/eliminate micro-contaminants of concern in line with the best available technology from a manufacturing perspective and encourages the development of new technology.

QCB occurs as a micro-contaminant in certain pest control products. PMRA's strategy with respect to QCB will likely mirror the steps followed for the risk management of hexachlorobenzene (HCB). These steps are likely to be:

- Identify all currently registered Technical Grade Active Ingredients containing QCB and estimate releases.
- Inform all registrants of these products and request action plans for the reduction of QCB, with the ultimate goal of virtual elimination.
- First round of requests will target the registrants of products containing 0.1 ppm or more (based on HCB strategy) of QCB and not already handled by another initiative.

Municipal Waste Incineration

QCB and TeCBs are unlikely to be found in the feed stream of municipal solid waste, and are therefore considered to be mainly products of incomplete combustion. Controlled incinerators using air pollution control systems designed to reduce mercury emissions through the use of powdered activated carbon should effectively control chlorobenzene emissions. Uncontrolled incinerators, such as conical and pit burners, are planned to be phased out by 2008 under the Canada-wide Standards for Dioxins and Furans (Canadian Council of Ministers of the Environment, 2001, 2003). This strategy would eliminate this source of chlorobenzenes.

Hazardous Waste Incineration

Implementation of the Canada-wide Standards for Dioxins and Furans will be monitored by a committee of the Canadian Council of Ministers of the Environment (CCME) responsible for air management. The implementation of the Canada-wide Standards is expected to favourably impact emissions of QCB and TeCBs. Since this strategy is already in place, no additional action is proposed for this sector.

Iron and Steel Plants

Canada-wide Standards for dioxins and furans (Canadian Council of Ministers of the Environment, 2003b) are already in place. No additional action is proposed for this sector.

Metals Processing

As the information for these sectors is incomplete, further investigation may be necessary before determining if action is needed in these sectors and what that action may be. As the estimated releases from this sector are relatively minor, developing an action plan for this sector is not a priority at this time.

Long-range Transport

QCB and TeCBs are not currently listed on either the United Nations Economic Commission for Europe Protocol on Persistent Organic Pollutants (UNECE POPs Protocol) or the Stockholm Convention on POPs (United Nations Environment Programme). Both agreements allow for the addition of new compounds. The UNECE ad hoc Scientific Expert Group on POPs is currently preparing information for the UNECE on substances requiring review under the Protocol and substances which may eventually be considered for addition. Countries are sharing the burden of carrying out this work. The Netherlands has prepared an information dossier on pentachlorobenzene for the ad hoc Scientific Expert Group on POPs. Information gathered by the Netherlands for the preliminary risk profile of pentachlorobenzene indicates that this substance meets the UNECE POP criteria for long-range transport, persistence, bioaccumulation toxicity and ecotoxicity.

Though the information dossier prepared by the Netherlands has not been put forward as a proposal to add pentachlorobenzene to the UNECE POPs protocol, it is expected that they will present this proposal in the future. At this point, pentachlorobenzene is not expected to be a high profile dossier, though the issue may be pushed by the Netherlands.

There is no immediate expectation that pentachlorobenzene will be added to the Stockholm Convention. It is anticipated, however, that substances added to the UNECE POPs protocol will eventually be considered for addition to the Stockholm Convention.

7.3 Follow-up

There are a number of existing initiatives addressing incidental releases in various sectors. Follow-up will be conducted on progress made under these initiatives. If it is determined that insufficient progress is being made through existing initiatives, Environment Canada will consider taking further action to address those sectors.

7.4 Virtual Elimination Plans

A virtual elimination plan is an information gathering tool which may be used as part of a long term strategy. At this point, it is not expected that virtual elimination plans will be requested. This action may be taken later on, if necessary, to evaluate progress towards virtual elimination.

8.0 BRIEF OUTLINE OF IMPLEMENTATION PLAN AND PROPOSED CONSULTATION APPROACH

8.1 QCB/TeCBs and Products Containing QCB/TeCBs

An on-line consultation approach was used to present this risk management strategy to stakeholders. The documents that were consulted on are available on Environment Canada's website.

The topics of discussion included:

- the planned addition of the substances to the proposed *Prohibition of Certain Toxic Substances Regulations, 2005*,
- the effect of the efforts currently in place regarding dioxins and furans and what their impact could be on chlorobenzene emissions.

9.0 NEXT STEPS / TIMELINE

Specific Risk Management Actions for Chlorobenzenes	Goal
Stakeholder consultations on Proposed Risk Management Strategy	Winter 2005
Publish revised risk management strategy on the Environment Canada website	Summer 2005
Publish the proposed <i>Regulations Amending the Prohibition of Certain Toxic Substances Regulations, 2005 (2-Methoxyethanol, Pentachlorobenzene and Tetrachlorobenzenes)</i> in the <i>Canada Gazette Part I</i>	Summer 2005
60-day public comment period on proposed Regulations	Summer 2005
Publish final regulations amending the <i>Prohibition of Certain Toxic Substances Regulations, 2005</i> in the <i>Canada Gazette Part II</i>	Summer 2006

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This report contains the collective views of an international group of experts and does not necessarily represent the decisions or the stated policy of the United Nations Environment Programme, the International Labour Organization, or the World Health Organization.

Concise International Chemical Assessment Document 60

**CHLOROENZENES OTHER THAN HEXACHLOROENZENE:
ENVIRONMENTAL ASPECTS**

First draft prepared by H.M. Malcolm, P.D. Howe, and S. Dobson, Centre for Ecology

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The **International Programme on Chemical Safety (IPCS)**, established in 1980, is a joint venture of the United Nations Environment Programme (UNEP), the International Labour Organization (ILO), and the World Health Organization (WHO). The overall objectives of the IPCS are to establish the scientific basis for assessment of the risk to human health and the environment from exposure to chemicals, through international peer review processes, as a prerequisite for the promotion of chemical safety, and to provide technical assistance in strengthening national capacities for the sound management of chemicals.

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FOREWORD

Concise International Chemical Assessment Documents (CICADs) are the latest in a family of publications from the International Programme on Chemical Safety (IPCS) — a cooperative programme of the World Health Organization (WHO), the International Labour Organization (ILO), and the United Nations Environment Programme (UNEP). CICADs join the Environmental Health Criteria documents (EHCs) as authoritative documents on the risk assessment of chemicals.

International Chemical Safety Cards on the relevant chemical(s) are attached at the end of the CICAD, to provide the reader with concise information on the protection of human health and on emergency action. They are produced in a separate peer-reviewed procedure at IPCS. They may be complemented by information from IPCS Poison Information Monographs (PIM), similarly produced separately from the CICAD process.

CICADs are concise documents that provide summaries of the relevant scientific information concerning the potential effects of chemicals upon human health and/or the environment. They are usually based on selected national or regional evaluation documents or on existing EHCs. Before acceptance for publication as CICADs by IPCS, these documents undergo extensive peer review by internationally selected experts to ensure their completeness, accuracy in the way in which the original data are represented, and the validity of the conclusions drawn.

The primary objective of CICADs is characterization of hazard and dose–response from exposure to a chemical. CICADs are not a summary of all available

data on a particular chemical; rather, they include only that information considered critical for characterization of the risk posed by the chemical. The critical studies are, however, presented in sufficient detail to support the conclusions drawn. For additional information, the reader should consult the identified source documents upon which the CICAD has been based.

Risks to human health and the environment will vary considerably depending upon the type and extent of exposure. Responsible authorities are strongly encouraged to characterize risk on the basis of locally measured or predicted exposure scenarios. To assist the reader, examples of exposure estimation and risk characterization are provided in CICADs, whenever possible. These examples cannot be considered as representing all possible exposure situations, but are provided as guidance only. The reader is referred to EHC 170.⁴

While every effort is made to ensure that CICADs represent the current status of knowledge, new information is being developed constantly. Unless otherwise stated, CICADs are based on a search of the scientific literature to the date shown in the executive summary. In the event that a reader becomes aware of new information that would change the conclusions drawn in a CICAD, the reader is requested to contact IPCS to inform it of the new information.

Procedures

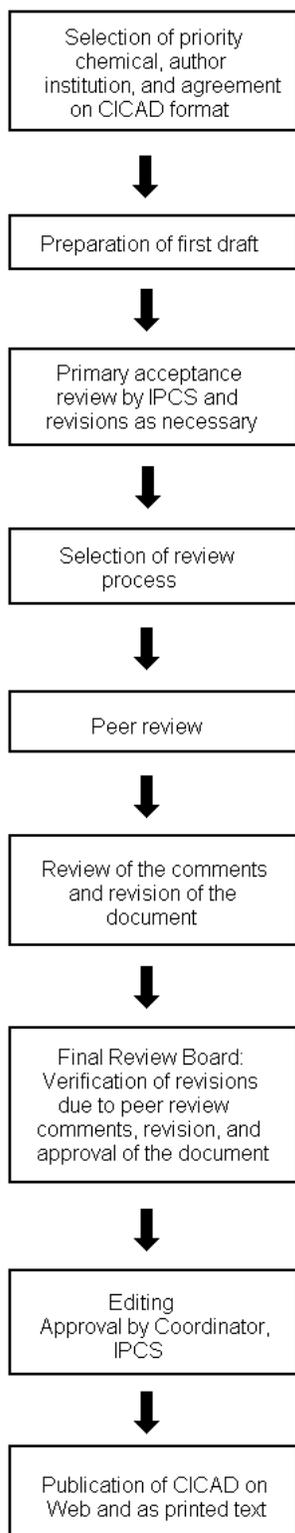
The flow chart on page 2 shows the procedures followed to produce a CICAD. These procedures are designed to take advantage of the expertise that exists around the world — expertise that is required to produce the high-quality evaluations of toxicological, exposure, and other data that are necessary for assessing risks to human health and/or the environment. The IPCS Risk Assessment Steering Group advises the Coordinator, IPCS, on the selection of chemicals for an IPCS risk assessment based on the following criteria:

- there is the probability of exposure; and/or
- there is significant toxicity/ecotoxicity.

Thus, it is typical of a priority chemical that

- it is of transboundary concern;
- it is of concern to a range of countries (developed, developing, and those with economies in transition) for possible risk management;
- there is significant international trade;
- it has high production volume;
- it has dispersive use.

CICAD PREPARATION FLOW CHART



Advice from Risk Assessment Steering Group

Criteria of priority:

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Thus, it is typical of a priority chemical that

- it is of transboundary concern;
- it is of concern to a range of countries (developed, developing, and those with economies in transition) for possible risk management;
- there is significant international trade;
- the production volume is high;
- the use is dispersive.

Special emphasis is placed on avoiding duplication of effort by WHO and other international organizations.

A prerequisite of the production of a CICAD is the availability of a recent high-quality national/regional risk assessment document = source document. The source document and the CICAD may be produced in parallel. If the source document does not contain an environmental section, this may be produced *de novo*, provided it is not controversial. If no source document is available, IPCS may produce a *de novo* risk assessment document if the cost is justified.

Depending on the complexity and extent of controversy of the issues involved, the steering group may advise on different levels of peer review:

- standard IPCS Contact Points
- above + specialized experts
- above + consultative group

The Steering Group will also advise IPCS on the appropriate form of the document (i.e., a standard CICAD or a *de novo* CICAD) and which institution bears the responsibility of the document production, as well as on the type and extent of the international peer review.

The first draft is usually based on an existing national, regional, or international review. When no appropriate source document is available, a CICAD may be produced *de novo*. Authors of the first draft are usually, but not necessarily, from the institution that developed the original review. A standard outline has been developed to encourage consistency in form. The first draft undergoes primary review by IPCS to ensure that it meets the specified criteria for CICADs.

The second stage involves international peer review by scientists known for their particular expertise and by scientists selected from an international roster compiled by IPCS through recommendations from IPCS national Contact Points and from IPCS Participating Institutions. Adequate time is allowed for the selected experts to undertake a thorough review. Authors are required to take reviewers' comments into account and revise their draft, if necessary. The resulting second draft is submitted to a Final Review Board together with the reviewers' comments. At any stage in the international review process, a consultative group may be necessary to address specific areas of the science. When a CICAD is prepared *de novo*, a consultative group is normally convened.

The CICAD Final Review Board has several important functions:

- to ensure that each CICAD has been subjected to an appropriate and thorough peer review;
- to verify that the peer reviewers' comments have been addressed appropriately;
- to provide guidance to those responsible for the preparation of CICADs on how to resolve any remaining issues if, in the opinion of the Board, the author has not adequately addressed all comments of the reviewers; and
- to approve CICADs as international assessments.

Board members serve in their personal capacity, not as representatives of any organization, government, or industry. They are selected because of their expertise in human and environmental toxicology or because of their experience in the regulation of chemicals. Boards are chosen according to the range of expertise required for a meeting and the need for balanced geographic representation.

Board members, authors, reviewers, consultants, and advisers who participate in the preparation of a CICAD are required to declare any real or potential conflict of interest in relation to the subjects under discussion at any stage of the process. Representatives of nongovernmental organizations may be invited to observe the proceedings of the Final Review Board. Observers may participate in Board discussions only at the invitation of the Chairperson, and they may not participate in the final decision-making process.

1. EXECUTIVE SUMMARY

This CICAD on chlorobenzenes other than hexachlorobenzene (environmental aspects) is an update of *Environmental Health Criteria (EHC) 128, Chlorobenzenes other than hexachlorobenzene* (IPCS, 1991a). Information on the fate and levels of chlorobenzenes was also obtained from Agency for Toxic Substances and Disease Registry reports on chlorobenzene (ATSDR, 1990) and 1,4-dichlorobenzene (ATSDR, 1998). A further literature search was performed up to December 2002 to identify any additional information published since these reviews were completed. Information on the peer review of the source document is presented in Appendix 1. Information on the peer review of this CICAD is presented in Appendix 2. This CICAD was approved as an international assessment at a meeting of the Final Review Board, held in Varna, Bulgaria, on 8–11 September 2003. Participants at the Final Review Board meeting are listed in Appendix 3. The International Chemical Safety Cards for a number of different chlorobenzenes (ICSC 0037, 0344, 0531, 0642, 0676, 1049, 1066, 1095, 1222), produced by the International Programme on Chemical Safety (IPCS, 2000, 2003a–h), have also been reproduced in this document. This CICAD concentrates on environmental aspects because there have been no significant changes to the human health assessment since publication of the EHC (IPCS, 1991a).

Chlorinated benzenes are a group of cyclic aromatic compounds in which one or more hydrogen atoms of the benzene ring have been replaced by a chlorine atom. Chlorobenzenes are used mainly as intermediates in the synthesis of pesticides and other chemicals. 1,4-Dichlorobenzene (1,4-DCB) is used in space deodorants and as a moth repellent. The higher chlorinated benzenes (trichlorobenzenes, 1,2,3,4-tetrachlorobenzene [1,2,3,4-TeCB], and pentachlorobenzene [PeCB]) have been used as components of dielectric fluids.

Natural sources of chlorobenzenes in the environment have not been identified. Chlorobenzenes are released to the environment during manufacture or use as intermediates in the production of other chemicals. They will also be released during the disposal of chlorobenzene products, such as from incinerators and hazardous waste sites. Monochlorobenzene (MCB) is released directly to the environment due to its use as a pesticide carrier. Chlorobenzenes used as deodorizers, fumigants, degreasers, insecticides, herbicides, and defoliants will also be released to the environment as a direct result of their application.

Their physicochemical properties suggest that chlorobenzenes released to the environment are likely to volatilize to the atmosphere. Removal of chlorobenzenes from the atmosphere will occur primarily via reactions with hydroxyl radicals to produce nitrochlorobenzene, chlorophenol, and aliphatic dicarbonyl products, which are further removed by photolysis or reaction with hydroxyl radicals. Chlorobenzenes released into the aquatic environment will be redistributed preferentially to the air and to sediment (particularly organically rich sediments). Chlorobenzenes in aqueous solutions could, in theory, undergo photochemical reductive dechlorination, although studies have been performed only under artificial conditions that were not representative of temperate regions. The most important factor affecting the behaviour and fate of chlorobenzenes in soil is sorption. Adsorption-desorption processes in soil affect the rate of volatilization and leaching and the availability of chemicals to microbial and chemical degradation or uptake by plants or other organisms.

Chlorobenzenes in various substrates, including soil, sediment, and sewage sludge, may be degraded by microorganisms. The major mechanism of aerobic degradation is via oxidative dechlorination, leading to the formation of hydroxylated aromatic compounds (mainly catechols), which undergo ring fission and subsequent mineralization to carbon dioxide and water. The less chlorinated benzenes are more readily degraded than the higher chlorinated ones.

The bioaccumulation of chlorobenzenes by aquatic organisms is determined by their relative water and lipid solubilities (thus reflecting the octanol/water partition coefficients) and the number of chlorine substitutions. Uptake from water increases with increasing chlorination and increasing temperature.

Concentrations of chlorobenzenes (MCB, dichlorobenzenes, and trichlorobenzenes) have been reported in ambient air, with mean concentrations in the order of 0.1 µg/m³ and maximum levels (at hazardous waste sites) of up to 100 µg/m³. Concentrations of chlorobenzenes in surface waters are generally in the ng/litre to µg/litre range, with maximum concentrations up to 0.2 mg/litre in areas close to industrial sources. Levels of chlorobenzenes in industrial wastewaters may be higher and vary according to the nature of the processes used. Chlorobenzene levels in uncontaminated soils are generally less than 0.4 mg/kg for dichlorobenzene congeners and less than 0.1 mg/kg for other chlorobenzene congeners. Levels of chlorobenzenes in sediments are generally in the ng/kg to µg/kg range, although levels in the mg/kg range have been reported in samples from industrial areas.

In general, aquatic toxicity increases with the degree of chlorination of the benzene ring. Seventy-two-hour EC₅₀s for green algae range from 5280 µg/litre for 1,3-DCB to 200 000 µg/litre for MCB; similarly, 48-h EC₅₀s for diatoms range from 8 to 235 000 µg/litre. For freshwater invertebrates, 48-h EC₅₀s range from 10 µg/litre for PeCB to >530 000 µg/litre for 1,2,4,5-TeCB. Ninety-six-hour LC₅₀s for fish range from 135 µg/litre for PeCB to 21 000 µg/litre for 1,2,4-trichlorobenzene (1,2,4-TCB). Chronic no-observed-effect concentrations (NOECs) for freshwater invertebrates range from 32 µg/litre for PeCB to 19 000 µg/litre for MCB; in fish, NOECs range from 18 µg/litre for PeCB to 8500 µg/litre for MCB.

Few data are available on the effects of chlorobenzenes on terrestrial systems. LC₅₀ values for plants grown hydroponically or in soil ranged from 0.028 to 9.3 mg/litre and from 1 to >1000 mg/kg soil, respectively. LC₅₀ values for the earthworms *Eisenia andrei* and *Lumbricus rubellus* ranged from 0.22 µmol/litre (pore water) for PeCB to 4281 µmol/litre for MCB.

The risk of chlorinated benzenes causing harm to aquatic organisms is low. Risk factors comparing chronic toxicity values with concentrations measured in the environment were generally below 1, with the exception of some compounds that had higher risk factors, with a maximum value of 200. The highest risk factors were derived using old data from point sources and are therefore unrepresentative of the whole environment, especially when the likelihood of evaporation is considered. There were inadequate data to perform a risk assessment for terrestrial species.

2. IDENTITY AND PHYSICAL/CHEMICAL PROPERTIES

Chlorinated benzenes are a group of cyclic aromatic compounds in which one or more hydrogen atoms of the benzene ring have been replaced by a chlorine atom. The generic molecular formula is C₆H_{6-n}Cl_n, where n = 1–6. There are 12 different chlorinated benzenes: monochlorobenzene (MCB), dichlorobenzene (DCB) (three isomers), trichlorobenzene (TCB) (three isomers), tetrachlorobenzene (TeCB) (three isomers), pentachlorobenzene (PeCB), and hexachlorobenzene. Hexachlorobenzene is reviewed in a separate EHC (IPCS, 1997) and is therefore not covered by this CICAD.

The identity of chlorobenzenes and their physical and chemical properties are presented in Table 1. MCB, 1,2-DCB, 1,3-DCB, and 1,2,4-TCB are colourless liquids, while all other congeners are white crystalline solids at room temperature. In general, the solubility of chlorobenzenes in water is low (decreasing with increasing chlorination), flammability is low, the octanol/water partition coefficients are moderate to high (increasing with increasing chlorination), and vapour pressures are low to moderate (decreasing with increasing chlorination) (IPCS, 1991a).

Table 1: Physicochemical properties of chlorobenzenes.^a

Chlorinated benzene	Abbreviation	CAS No.	Molecular formula	Relative molecular mass	Melting point (°C)	Boiling point ^b (°C)	Vapour pressure at 25 °C (Pa)	Aqueous solubility at 25 °C (mg/litre)	Henry's law constant (kPa·m ³ /mol)	Log octanol/water partition coefficient (K _{ow})	Soil sorption coefficient (K _{oc})
Monochlorobenzene	MCB	108-90-7	C ₆ H ₅ Cl	112.6	-45.6	132.0	1665	293	0.377	2.98	466
1,2-Dichlorobenzene	1,2-DCB	95-50-1	C ₆ H ₄ Cl ₂	147.0	-17.0	180.5	197	91.1	0.198	3.38	987
1,3-Dichlorobenzene	1,3-DCB	541-73-1	C ₆ H ₄ Cl ₂	147.0	-24.7	173.0	269	123	0.366	3.48	1070

1,4-Dichlorobenzene	1,4-DCB	106-46-7	C ₆ H ₄ Cl ₂	147.0	53.1	174.0	90	30.9	0.160	3.38	1470
1,2,3-Trichlorobenzene	1,2,3-TCB	87-61-6	C ₆ H ₃ Cl ₃	181.5	53.5	218.5	17.3	12.2	0.306	4.04	3680
1,2,4-Trichlorobenzene	1,2,4-TCB	120-82-1	C ₆ H ₃ Cl ₃	181.5	17.0	213.5	45.3	45.3	0.439	3.98	2670
1,3,5-Trichlorobenzene	1,3,5-TCB	108-70-3	C ₆ H ₃ Cl ₃	181.5	63.5	208.0	24.0	3.99	0.233	4.02	NA ^c
1,2,3,4-Tetrachlorobenzene	1,2,3,4-TeCB	634-66-2	C ₆ H ₂ Cl ₄	215.9	47.5	254.0	5.2	12.1	0.261	4.55	NA
1,2,3,5-Tetrachlorobenzene	1,2,3,5-TeCB	634-90-2	C ₆ H ₂ Cl ₄	215.9	54.5	246.0	9.8	2.81	0.593	4.65	8560
1,2,4,5-Tetrachlorobenzene	1,2,4,5-TeCB	95-94-3	C ₆ H ₂ Cl ₄	215.9	139.5	243.6	0.72	2.16	0.261	4.51	6990
Pentachlorobenzene	PeCB	608-93-5	C ₆ HCl ₅	250.3	86.0	277.0	133 ^d	0.83	0.977	5.03	58 700

a From IPCS (1991a).

b Calculated at atmospheric pressure (101.3 kPa), except for 1,3,5-TCB, which was at 93.5 kPa.

c NA = not available.

d Calculated at 98 °C.

3. ANALYTICAL METHODS

The analytical technique of choice for the determination of chlorobenzenes in environmental samples is gas chromatography (GC). However, the methods of collection and preparation of samples for GC analysis vary considerably, depending on the medium and the laboratory. Capillary columns with different stationary phases are frequently used to separate compounds. Detection occurs via the use of a flame ionization detector (FID), electron capture detector (ECD), or mass spectrometric (MS) detector (IPCS, 1991a).

Tenax-GC resins have commonly been used as adsorbents for the air sampling of chlorobenzenes (Krost et al., 1982; Pellizzari et al., 1982), although XAD resins have also been used (Langhorst & Nestruck, 1979). Air pollutants collected on Tenax-GC resins can be desorbed directly onto the GC column by heating the tube with sorbent. XAD resins can be extracted with solvents, an aliquot of which can then be injected into a GC. Detection limits in the 1970s ranged from 0.7 µg/m³ for MCB to 0.9 µg/m³ for PeCB (Langhorst & Nestruck, 1979); however, much lower detection limits have been achieved more recently using ECD (0.5 pg/m³ for PeCB to 1.8 pg/m³ for 1,2,4,5-TeCB) (Hermanson et al., 1997).

Solvent extraction is a simple and effective technique for recovering chlorobenzenes from water samples. Hexane, pentane, and a 1:1 mixture of cyclohexane and diethyl ether have been identified as suitable extraction solvents for these compounds (Oliver & Bothen, 1980; Piet et al., 1980; Otson & Williams, 1981; Meharg et al., 2000). Alternatively, preconcentration of the chlorobenzenes on organic resins, such as Chromosorb 102 and Tenax-GC, is also effective; detection limits using Chromosorb 102 were reported to range from 0.5 µg/litre for MCB to 0.01 ng/litre for PeCB (Oliver & Bothen, 1980; Pankow & Isabelle, 1982). The purge-and-trap method has also been used to concentrate the volatile halogenated benzenes before analysis using GC (Jungclaus et al., 1978; Pereira & Hughes, 1980; Otson & Williams, 1982; Huybrechts et al., 2000; Martinez et al., 2002). Detection limits of 0.1–0.2 µg/litre for MCB and various dichlorobenzene isomers were achieved using FID and Hall electrolyte conductivity detectors (Otson & Williams, 1982), 0.08 µg/litre for 1,2,4-TCB using ECD (Martinez et al., 2002), and 0.76–20 ng/litre for di- and trichlorobenzenes using MS (Huybrechts et al., 2000). More recently, alternative extraction techniques such as headspace solid-phase microextraction with GC-MS have achieved detection limits for individual chlorobenzene isomers ranging from 4 to 6 ng/litre (He et al., 2000); however, it should be noted that analytical techniques using simple solvent extraction and GC-MS can now attain detection limits ranging from 5 pg/litre for 1,2,3- and 1,3,5-TCB to 15 pg/litre for PeCB (Meharg et al., 2000).

The extraction of chlorobenzenes from aquatic sediments, sewage sludges, or soil can be achieved by solvent or Soxhlet extraction (Oliver & Bothen, 1982; Lopez-Avila et al., 1983; Onuska & Terry, 1985; Wang & Jones, 1991; Wang et al., 1992). Solvents commonly used are acetone and/or hexane. Other extraction methods, such as sonication, saponification, and supercritical fluid extraction, have been used to extract sediment-bound chlorobenzenes, but were found to be less efficient than Soxhlet extraction (Prytula & Pavlostathis, 1996). The extract is generally dried using sodium sulfate, followed by cleanup on a Florisil column before GC analysis with ECD, with detection limits of 1500 µg/kg for MCB and lower detection limits ranging from 1.5 µg/kg for dichlorobenzenes to 0.05 µg/kg for PeCB (Oliver & Bothen, 1982; Onuska & Terry, 1985; Wang & Jones, 1991; Wang et al., 1992). Alternatively, headspace solid-phase microextraction with GC-ion trap MS has been found to reproduce detection limits of 0.03–0.1 µg/kg for 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB in soil (Santos et al., 1997).

For the detection of chlorobenzenes in biota samples, solvent or Soxhlet extraction with subsequent cleanup on Florisil columns and GC analysis with ECD have commonly been used (Lunde & Ofstad, 1976; Kuehl et al., 1980; Oliver & Bothen, 1982; Muir et al., 1992; Gebauer & Weseloh, 1993; Cobb et al., 1994; Jan et al., 1994; Wade et al., 1998). Detection limits of 1500 µg/kg for MCB and lower detection limits ranging from 5 µg/kg for dichlorobenzenes to 0.02 µg/kg for PeCB have been reported (Oliver & Bothen, 1982; Cobb et al., 1994). Vacuum extraction and the direct purge-and-trap method have also been used to quantify levels of MCB in fish tissue (Hiatt, 1981).

4. SOURCES OF ENVIRONMENTAL EXPOSURE

4.1 Natural sources

Natural sources of chlorobenzenes in the environment have not been identified. However, 1,2,3,4-TeCB has been identified in the oil of marsh grass, although it is not known whether this was formed naturally (Miles et al., 1973).

4.2 Anthropogenic sources

Chlorobenzenes are released to the environment from sites where they are either manufactured or used as intermediates in the production of other

chemicals. They will also be released during the disposal of chlorobenzene products, such as from incinerators (IPCS, 1991a) and hazardous waste sites (ATSDR, 1998). Chlorobenzenes are a product of incomplete combustion and may therefore be released to the environment from waste incinerators. Chlorobenzenes may be formed from the metabolic breakdown of lindane in higher organisms and from its physical breakdown under extreme environmental conditions (IPCS, 1991b).

Releases of some chlorobenzene compounds to the environment in the USA in 2001, as recorded in the US Toxics Release Inventory (TRI), are listed in Table 2. These data do not form a comprehensive list, as only certain types of industrial facility are required to register in the TRI (ATSDR, 1998). There is a paucity of data on the quantity of chlorobenzenes released to the environment in other parts of the world, although some production and consumption data are available. Approximately 15 000 tonnes of 1,4-DCB were produced in and/or imported into the European Union in 1994 (EC, 2001). Total production of MCB, 1,2-DCB, and 1,4-DCB in Japan in 1998 was 26 351 tonnes (Chemical Daily Company, 1999), with 9073 tonnes imported in 1998 and 8310 tonnes imported in 1999 (Chemical Daily Company, 2000).

Table 2: Total releases of chlorobenzenes in the USA during 2001.^a

	Releases (tonnes)					
	MCB	1,2-DCB	1,3-DCB	1,4-DCB	1,2,4-TCB	PeCB
Total emissions to air	314	56	0.50	37	43.92	0.03
Surface water discharges	0.3	0.38	0.26	0.51	0.04	0.06
Releases to land	0.01	0.00	0.00	0.00	3.5	1.07
Total on-site releases	362	59	0.76	42	49	1.16
Total off-site releases	2.5	0.52	0.46	0.69	4.2	0.09

^a From US EPA (2003).

Some uses of chlorobenzenes, including uses as deodorizers, fumigants, degreasers, insecticides, herbicides, and defoliants, will result in direct releases to the environment.

MCB will be released directly to the environment due to its use as a pesticide carrier (Meek et al., 1994c). MCB is used as a solvent carrier for pesticides (29 000 kg per annum in Canada), in the manufacture of rubber polymers (20 000 kg per annum in Canada), and as a carrier for textile dyes (1000 kg per annum in Canada) (Mackay et al., 1996). Fifty per cent of the MCB used in Canada is released to the environment; 80% is emitted to the atmosphere, 10% to water, and 10% to soil, giving releases of 20 000, 2500, and 2500 tonnes, respectively, per year (Mackay et al., 1996). MCB is used in the production of phenol and nitrochlorobenzene (*ortho* and *para* isomers), in the formulation of herbicides, to produce additional chlorobenzenes, and as a solvent in the manufacture of adhesives, paints, resins, dyestuffs, and drugs (Grosjean, 1991). MCB is used in the manufacture of diphenyl oxide, phenylphenol, silicone resin, and other halogenated organics (ATSDR, 1990).

1,2-DCB is used primarily in the automotive and metal industries as a solvent for the removal of carbon and degreasing of metal parts (Meek et al., 1994a). 1,2-DCB is used in the synthesis of organic chemicals such as toluene diisocyanate (Grosjean, 1991).

1,4-DCB is used in air fresheners, urinal deodorants, and moth and bird repellents (Meek et al., 1994b; EC, 2001). All of these uses release 1,4-DCB to the environment, principally the atmosphere. 1,4-DCB is also used as an intermediate in the production of other chemicals, including polyphenylene sulfide resins (Grosjean, 1991) and 1,2,4-TCB (ATSDR, 1998). Minor uses of 1,4-DCB include its use in the control of tree-boring insects, ants, and blue mould in tobacco seedbeds (ATSDR, 1998).

Trichlorobenzenes, especially 1,2,4-TCB, are used as dye carriers, degreasing solvents, oil additives, and dielectric fluids and in the formulation of pesticides (Grosjean, 1991). The use of trichlorobenzenes is restricted to mainly 1,2,4-TCB, which is used as a chemical intermediate and an industrial solvent (Giddings et al., 1994c). 1,2,4-TCB was formerly used as a degreasing agent, in septic tanks, and in drain cleaners, wood preservatives, and abrasive formulations (EC, 2003).

Tetrachlorobenzenes and pentachlorobenzenes may be released to the environment from the spillage of dielectric fluids (Giddings et al., 1994a,b). 1,2,3,4-TeCB is used as a component in dielectric fluids (IPCS, 1991a). 1,2,4,5-TeCB is used as an intermediate in the manufacture of herbicides and defoliants. It is also used as an insecticide, as a moisture-resistant impregnant, in electrical insulation, and in packing protection (IPCS, 1991a). PeCB was formerly used in a pesticide to combat oyster drills (small snails that eat oysters). It has also been used as an intermediate (IPCS, 1991a).

5. ENVIRONMENTAL TRANSPORT, DISTRIBUTION, AND TRANSFORMATION

5.1 Transport and distribution

Their physicochemical properties suggest that chlorobenzenes released to the environment are likely to be volatilized to the atmosphere. The Henry's law constants measured for chlorobenzenes suggest that they are readily volatilized, especially from aquatic systems with long residence times, such as large lakes and oceans (Ten Hulscher et al., 1992). However, chlorobenzenes released to water may also be adsorbed onto sediment, especially if it is rich in organic matter. Volatilization from soil is also likely, although, depending on the characteristics of the soil, there may also be sorption to soil.

The majority of chlorobenzenes added to soil, as either sewage sludge or spiked samples, were volatilized, with biodegradation and abiotic degradation insignificant compared with the amount volatilized (Wang & Jones, 1994a). Volatilization occurred by two-step first-order processes, with high rates of volatilization during an initial step, followed by a second, much slower step, which was presumably controlled by the rate of desorption of the compound from soil. Half-lives for loss of chlorobenzenes ranged from 13.0 to 219 days for sewage sludge applications and from 10.6 to 103 days for spiked samples. Half-lives increased with increasing chlorination and were also higher in sludge-amended soil than in the spiked samples. The half-lives for volatilization of MCB and 1,2-DCB from soil were 2.1 and 4.0 days, respectively. Initial soil concentrations were 100 mg/kg dry weight (Anderson et al., 1991). Transient

geochemical conditions can significantly alter the extent of removal. Robertson (1994) studied the fate of a dichlorobenzene mixture (containing 74% 1,2-DCB, 11% 1,3-DCB, and 15% 1,4-DCB) released to sub-surface soil in effluent from a septic system. High dichlorobenzene concentrations were found in the aerobic unsaturated zone (below the septic system) where dichlorobenzene had a residence time of 60 days. The migration of dichlorobenzene to the water table was attenuated by this zone.

The most important factor affecting the behaviour and fate of chlorobenzenes in soil is sorption. Adsorption-desorption processes in soil affect the rate of volatilization and leaching and the availability of chemicals to microbial and chemical degradation or uptake by plants or other organisms (Wang & Jones, 1994a). The soil sorption coefficients for chlorobenzenes range from 466 to 58 700 (Table 1) and generally increase with increasing chlorination (IPCS, 1991a; Schrap et al., 1994). Sorption of chlorobenzenes to soil is affected by many parameters, and it increases with increasing organic matter content (Barber et al., 1992; Faschan et al., 1993).

The adsorption of 1,2,4-TCB to soil was found to decrease with increasing soil depth (Njoroge et al., 1998). These depth-related changes were attributed to changes in composition, texture, and accessibility of the soil organic matter. At deeper levels, extractable organic matter was increasingly dominated by fulvic acids. The higher fulvic:humic acid ratio in deep soil reflects an increasing hydrophilicity of the soil organic matter. Abundance of iron oxide and size of clay particles also increase with depth.

Sorption of chlorobenzenes is also affected by soil moisture, with reduced sorption to wet soil (Chiou & Shoup, 1985; Thibaud et al., 1993). Adsorption of 1,2,4-TCB to soil was reduced following the addition of sodium dodecyl sulfate, a surfactant that frequently occurs in sewage sludge disposed of onto land (DiVincenzo & Dentel, 1996). Desorption occurred only when the sodium dodecyl sulfate concentration exceeded the critical micelle concentration. However, increased adsorption of MCB to soil was reported following the addition of the surfactant hexadecyltrimethylammonium (HDTMA) (Sheng et al., 1998). Adsorption on the HDTMA phase was 80–160 times higher than sorption on natural organic matter. The sorption of 1,4-DCB by aquifer materials with a low organic carbon content was enhanced in the presence of tetrachloroethene (Brusseau, 1991). The enhanced sorption was suggested to arise from tetrachloroethene increasing the organic carbon content of the sorbent.

Desorption of 1,3-DCB from a silty soil to deionized water had an initial fast labile phase, followed by a slow phase (Lee et al., 2002). An average of 60% of the initial concentration was desorbed. The first-order rate constant was 0.022–0.038 per hour for the labile phase and 4.1×10^{-5} to 7.8×10^{-4} per hour for the slow desorption phase. Single-step batch tests showed that desorption of chlorobenzenes from sediment was slow, with less than 0.5% of 1,2,4,5-TeCB and PeCB desorbed within 62 days. Desorption of 1,2,4-TCB was significantly higher than that of other compounds, with 3% desorbed within 62 days (Gess & Pavlostathis, 1997).

MCB adsorbed onto marine sediment reached equilibrium within 3 h (Zhao et al., 2001). Equilibrium took the same time in natural seawater, artificial seawater, and deionized water. Adsorption occurred via the surface and micropores of sediment and could be described by either the Freundlich or Langmuir model. Adsorption was not affected by temperature (18, 25, or 30 °C), although the saturate adsorption amount decreased at higher temperatures. Adsorption isotherms and the saturate adsorption amounts were higher in natural seawater than in artificial seawater and deionized water. Adsorption of 1,2,4,5-TeCB on sandy aquifer solids took up to hundreds of days to reach equilibrium (Ball & Roberts, 1991). Distribution coefficients were greatest in the size fraction with the largest grains.

Mean (\pm SD) suspended sediment/water partition coefficients ($\log K_{oc}$) for chlorobenzenes measured in Ise Bay, Japan, were 3.47 ± 0.74 (1,3-DCB), 3.69 ± 0.48 (1,2-DCB), 3.61 ± 0.39 (1,2,3-TCB), 3.86 ± 0.40 (1,2,4-TCB), 3.55 ± 0.47 (1,3,5-TCB), 4.39 ± 0.33 (1,2,3,4-TeCB), 3.94 ± 0.33 (1,2,3,5-TeCB and 1,2,4,5-TeCB), and 4.59 ± 0.41 (PeCB) (Masunaga et al., 1996b). Concentrations of chlorobenzenes in water and adsorbed onto suspended sediment were compared. None of the chlorobenzenes gave a clear adsorbed level distribution pattern, and the correlation between soluble and adsorbed chlorobenzenes was weak.

The fate of MCB, 1,2-DCB, and 1,2,4-TCB in wastewater applied to soil was examined in a microcosm experiment (Piwoni et al., 1986). Initial concentrations of MCB, 1,2-DCB, and 1,2,4-TCB in the wastewater were 1.9–3.1, 2.4, and 0.72 $\mu\text{mol/litre}$, respectively. The proportions of MCB and 1,2-DCB volatilized were 14% and 21%, respectively, and it was assumed that 84% and 79%, respectively, were degraded, giving concentrations in the volume effluent of $9 \pm 10\%$ of the original concentration. Volatility of 1,2,4-TCB was not measured, but it was assumed to be approximately 89%, as $<0.7\%$ of the original concentration remained in the effluent.

The half-life for dichlorobenzene (all isomers) in a septic groundwater system was 15 days (Robertson, 1994). The site included a 2-m-thick, sandy aerobic unsaturated zone. This loss was due to a combination of volatilization and aerobic biodegradation. Biodegradation occurred after an initial lag phase and was most likely for 1,3-DCB and 1,4-DCB. Dichlorobenzene in the anaerobic zone was not readily biodegraded.

Octanol/air partition coefficients ($\log K_{oa}$) measured for chlorobenzenes at 25 °C were 4.36 (1,2-DCB), 5.19 (1,2,3-TCB), 5.64 (1,2,3,4-TeCB), 5.63 (1,2,4,5-TeCB), and 6.27 (PeCB) (Harner & Mackay, 1995). Octanol/air partition coefficients determined partitioning from the atmosphere to vegetation, soils, and possibly aerosols.

Microcosm experiments suggested that 1,2-DCB in soil was not taken up by grass (*Holcus lanatus*) roots, although some foliar adsorption of dichlorobenzene volatilized from soil was reported (Wilson & Meharg, 1999). A root concentration factor of 19 litres/kg has been reported for 1,2,4-TCB (Dietz & Schnoor, 2001). From these data, it can be assumed that tri- and/or tetrachlorinated benzenes have the potential to be taken up by plants.

5.2 Transformation

5.2.1 Abiotic degradation

Removal of chlorobenzenes from the atmosphere will occur primarily via reactions with hydroxyl radicals to produce nitrochlorobenzene, chlorophenol, and aliphatic dicarbonyl products, which are further removed by photolysis or reaction with hydroxyl radicals. Photolysis and reactions with ozone or nitrate radicals are of negligible importance (Grosjean, 1991). Rate constants for reactions with hydroxyl radicals (in cm^3/s per molecule) were calculated to be 8.8×10^{-13} (MCB), 4.0×10^{-13} (1,2-DCB), 7.2×10^{-13} (1,3-DCB), 4.3×10^{-13} (1,4-DCB), 6.0×10^{-13} (1,2,3-TCB), and 5.65×10^{-13} (1,2,4-TCB) (Atkinson et al., 1985; Klöpffer et al., 1986; Dilling et al., 1988; Arnts et al., 1989). A rate constant for reaction of MCB with ozone was calculated to be $<5 \times 10^{-21}$ cm^3/s per molecule. Assuming 24-h average hydroxyl radical and ozone concentrations of 1×10^6 and 7.2×10^{11} molecules/ cm^3 , tropospheric half-lives for MCB were calculated to be 13 days for reactions with hydroxyl radicals and >8.8 years for reactions with ozone (Atkinson et al., 1985). Tropospheric half-lives for 1,4-DCB and 1,2,4-TCB reacting with hydroxyl radicals were calculated to be 33.4 and 26.7 days, respectively (Klöpffer et al., 1988).

1,2,4-TCB in the atmosphere may be degraded via direct photolysis, although this route of degradation is minor, due to the poor spectral overlap between the solar spectrum and the adsorption spectrum of 1,2,4-TCB. The maximum photolysis rate for 1,2,4-TCB in summer at midday under clear skies was 0.03% per hour (Bunce et al., 1989).

Chlorobenzenes in aqueous solutions may undergo photochemical reductive dechlorination. PeCB was degraded to tetrachlorobenzenes, which in turn were photodegraded to trichlorobenzenes, dichlorobenzenes, MCB, and, ultimately, phenol, benzene, and hydrogen chloride (Chu & Jafvert, 1994). These reactions were reported following exposure to 253.7-nm monochromatic ultraviolet lamps. The rate of photodegradation increased in the presence of surfactants. In addition to the main reductive pathway of photodechlorination, minor pathways, including photochlorination, photohydrolysis, and photoisomerization, also occurred. 1,2,3,5-TeCB was photolysed to 1,2,4-TCB or 1,3,5-TCB in the presence of an acetone sensitizer (Choudhry & Hutzinger, 1984). Photochemical reactions in the absence of a sensitizer transformed tetrachlorobenzenes into other isomers and also produced some chlorobenzenes with greater chlorination than the original tetrachlorobenzene compound. The rate constant for reaction of 1,2,4-TCB with hydroxyl radicals in an acidic solution was $6.0 \pm 0.3 \times 10^9$ per mol/litre per second (Gallard & De Laat, 2001). 1,4-DCB in aqueous solution was photodegraded to 4-chlorophenol, hydroquinone, hydroxybenzoquinone, and 2,5-dichlorophenol (Meunier et al., 2001). The formation of 2,5-dichlorophenol demonstrates hydroxylation without dechlorination. Photodegradation of MCB in aqueous solutions has been reported under both aerobic and anaerobic conditions and at pHs ranging from <1 to <12 (Tissot et al., 1983, 1984; Dilmeghani & Zahir, 2001). Degradation followed first-order kinetics, with rate constants ranging from 1.8×10^{-4} to 6.4×10^{-4} per second for anaerobic and oxygen-saturated conditions, respectively (Dilmeghani & Zahir, 2001). The rate of degradation was an order of magnitude higher with ultraviolet and hydrogen peroxide or hydrogen peroxide–ozone compared with ultraviolet alone.

The half-lives for photolytic degradation of MCB and 1,2,4-TCB in surface water, simulating summer conditions at 40 degrees latitude, were 170 and 450 years, respectively (Dulin et al., 1986).

5.2.2 Biodegradation

Chlorobenzenes in various substrates, including soil, sediment, and sewage sludge, can be degraded by microorganisms. The major mechanism of aerobic degradation is via oxidative dechlorination, usually initiated by dioxygenative hydroxylation, leading to the formation of hydroxylated aromatic compounds (mainly catechols), which undergo ring fission and subsequent mineralization to carbon dioxide and water. The less chlorinated benzenes are more readily degraded than the higher chlorinated ones (IPCS, 1991a). Biodegradation under anaerobic conditions has also been reported, although this occurs at a slower rate than aerobic biodegradation.

Chlorobenzene-degrading bacteria isolated from aerobic environments include *Burkholderia* (previously known as *Pseudomonas*) species (strains JS150, P51, JS6, PS12, and PS14) (Pettigrew et al., 1991; Sander et al., 1991, 1997; Van der Meer et al., 1991, 1997; Nishino et al., 1994; Beil et al., 1997; Meckenstock et al., 1998), *Alcaligenes* species (strains A175 and OBB65) (De Bont et al., 1986; Schraa et al., 1986), *Escherichia hermannii* (Kiernicka et al., 1999), *Nitrosomonas europaea* (Keener & Arp, 1994), *Mycobacterium vaccae*, and *Rhodococcus* species (strain R22) (Fairlee et al., 1997).

The degradative abilities of these bacteria vary, with some organisms exhibiting a lag or adaptation period prior to degradation. Some can degrade several chlorobenzenes (Brunsbach & Reineke, 1994), whereas others are compound-specific (Reineke & Knackmuss, 1984; Brunsbach & Reineke, 1994; Keener & Arp, 1994). For some, degradation occurs only in the presence of other sources of carbon and energy, whereas others are able to use chlorobenzenes as their sole carbon and energy source (Van der Meer et al., 1987). Genetic analysis has shown that these bacteria contain a novel combination of previously existing genes — genes for aromatic ring dioxygenase and dihydrodiol dehydrogenase — and other genes for a chlorocatechol oxidative pathway.

Degradation is also dependent upon the initial chlorinated benzene concentrations. Degradation will occur only if the initial concentration is below the toxic threshold, although bacteria that have previously been exposed to MCB have the ability to degrade higher concentrations than those that did not have prior exposure. For example, concentrations of MCB greater than 2.5 mmol/litre (282 mg/litre) were found to be toxic to *Pseudomonas* sp. strain RHO1 cells. Cells that had previously been exposed to MCB demonstrated toxicity at concentrations greater than 3.5 mmol/litre (394 mg/litre) (Fritz et al., 1992).

MCB and 1,2,4-TCB were degraded by bacteria isolated from solids sampled from pristine aquifers (Swindoll et al., 1988). Degradation followed first-order rate constants, with V_{\max} values of 0.38–2.71 ng/g per hour for 1,2,4-TCB. Degradation of MCB was not saturated; therefore, V_{\max} could not be calculated.

A consortium of Gram-negative and Gram-positive bacteria isolated from groundwater and soil contaminated with MCB was able to mineralize 54% of a 2.23 $\mu\text{mol/litre}$ solution via the modified *ortho* pathway within 7 days in the presence of nutrients. Degradation also occurred without added nutrients, although at a slower rate (Nishino et al., 1992).

Degradation of 1,4-DCB occurred at similar rates under aerobic or anaerobic conditions and was enhanced in mixtures with high sludge content (which reduced overall oxygen) (Gejlsbjerg et al., 2001). Mineralization occurred after a lag phase of 30 days. After inoculation for 2 months, mineralization was 12.4% in the sludge and 21.6% in the 1:20 sludge:soil mixture. The authors concluded that mineralization was probably occurring in the aerobic layers of the sludge–soil mixtures, as mineralization did not occur in sludge in the absence of molecular oxygen.

A consortium of bacteria isolated from Rhine sediment was able to degrade PeCB, 1,2,3,4-TeCB, 1,2,3,5-TeCB, 1,2,4,5-TeCB, and 1,2,3-TCB via reductive dechlorination in the presence of lactate, glucose, ethanol, or isopropanol as the electron donor (Holliger et al., 1992). PeCB was degraded to 1,3,5-TCB, while 1,2,3,4-TeCB and 1,2,4,5-TeCB were degraded to 1,2,4-TCB. Chlorobenzenes that were not dechlorinated during the 4-week incubation included 1,2,4-TCB, 1,3,5-TCB, and all isomers of dichlorobenzene. Other studies have reported complete mineralization of some higher chlorinated compounds. Two *Pseudomonas* strains (PS12 and PS14) isolated from the soil of an industrial waste deposit were able to mineralize various chlorobenzenes, including MCB, all three dichlorobenzenes, 1,2,4-TCB, and 1,2,4,5-TeCB (strain PS14 only). 1,2,4-TCB and 1,2,4,5-TeCB were degraded via dioxygenation of the aromatic ring, producing 3,4,6-trichlorocatechol. Subsequent *ortho* cleavage, catalysed by a Type II catechol 1,2-dioxygenase, produced 2,3,5-trichloromuconate, which was degraded via the tricarboxylic acid pathway (Sander et al., 1991).

Degradation of 1,2-DCB and 1,4-DCB within a mixture of organic compounds was reported in a 149-day batch microcosm using sediment and groundwater obtained from various sampling sites of an aquifer (Nielsen & Christensen, 1994). The initial concentrations were approximately 120 $\mu\text{g/litre}$. Within an average of 82 days, 78.3% of 1,4-DCB and 81.0% of 1,2-DCB were degraded. The lag phases were 4.9 and 4.5 days for 1,4-DCB and 1,2-DCB, respectively. In the Organisation for Economic Co-operation and Development (OECD) closed bottle test, 67% of an initial 1,4-DCB concentration of 1.9 mg/litre was mineralized after 28 days, indicating that 1,4-DCB is readily degradable (Topping, 1987).

Bartholomew & Pfaender (1983) calculated degradation rates for MCB and 1,2,4-TCB at different sites of a river system during different seasons. Rates of degradation of MCB and 1,2,4-TCB were reported to decrease over the freshwater to estuarine to marine gradient. V_{\max} values for MCB degradation during

May and September were 13–14 ng/litre per hour for fresh water, 4.9–10 ng/litre per hour for estuarine water, and <1–1.7 ng/litre per hour for marine water. V_{\max} values were <1 ng/litre per hour at all three sites in February. The corresponding values for degradation of 1,2,4-TCB in May and July were <1–7.5 ng/litre per hour for fresh water, <1–7.9 ng/litre per hour for estuarine water, and <1–2.3 ng/litre per hour for marine water.

In controlled lysimeter experiments, 80% of 1,2,4,5-TeCB in soils and liquid cultures was mineralized by the bacterial strains *Isphingomonas* sp. strains HH69 and RW1 and *Pseudomonas* sp. strain PS14 within a few days (Figge et al., 1993). Degradation was not increased in the presence of additional energy sources such as peptone, triolein, and glucose. Degradation did not occur in acidic soils (pH < 4).

Biodegradation of chlorobenzenes has also been reported in several studies under anaerobic conditions, including methanogenic and sulfate-reducing conditions. As with aerobic degradation, degradability varies between organisms. Under anaerobic conditions, degradation is limited to dechlorination, with no breakdown of the aromatic structure.

Anaerobic degradation of chlorobenzenes has been reported in river sediment (Masunaga et al., 1996a; Susarla et al., 1996). Dechlorination occurred without a lag period, with half-lives ranging from 17 to 433 days. The main pathway for PeCB dechlorination was via 1,2,4,5-TeCB, 1,2,4-TCB, 1,4-DCB, and MCB. A minor pathway, via 1,2,3,4-TeCB, 1,2,3-TCB, 1,2-DCB, and MCB, was also observed. MCB was stable under anaerobic conditions. The preferences for dechlorination were two adjacent chlorine atoms, followed by one chlorine on an adjacent carbon, followed by no chlorine on the adjacent carbon. Other studies have reported similar anaerobic biodegradation (Beurskens et al., 1991; Ramanand et al., 1993; Susarla et al., 1997). Nowak et al. (1996) reported anaerobic degradation of all chlorobenzenes, including MCB, to benzene.

In anaerobic sewage sludge, PeCB was dechlorinated to 1,2,3,4-TeCB and 1,2,3,5-TeCB, which were degraded to 1,2,4-TCB, 1,2,3-TCB, and 1,3,5-TCB, and then 1,2-DCB and 1,3-DCB (Yuan et al., 1999). Sequential dechlorination occurred within a substrate concentration range of 2–50 mg/litre, but was slower at concentrations greater than 50 mg/litre. Dechlorination rates were highest under methanogenic conditions (0.30 mg/litre per day), with slower rates under sulfate-reducing (0.12 mg/litre per day) and denitrifying conditions (0.08 mg/litre per day). The rate of dechlorination of 1,2,3-TCB by anaerobic sediment ranged from 15 to 35 pmol/ml wet sediment per day (Yonezawa et al., 1994).

Some studies have shown chlorobenzenes to be resistant to anaerobic biodegradation. Nielsen et al. (1995) reported no biodegradation of 1,2-DCB or 1,4-DCB in anaerobic landfill leachate collected from four different sites at distances ranging from 2 to 350 m from the landfill. The governing reactions, which varied at each site, included methanogenesis, iron(III) reduction, nitrate reduction, and manganese(IV) reduction. Dichlorobenzenes have been reported to persist for at least 20 years in an aquifer that had been contaminated with rapid-infiltration sewage disposal (Barber, 1988). 1,2,3,5-TeCB and 1,3,5-TCB were resistant to degradation by soil slurry microorganisms that could degrade PeCB, 1,2,3,4-TeCB, and 1,2,4-TCB (Ramanand et al., 1993).

5.3 Bioaccumulation

The bioaccumulation of chlorobenzenes by aquatic organisms is determined by their relative water and lipid solubility (thus reflecting the octanol/water partition coefficients) and the number of chlorine substitutions. Uptake from water increases with increasing chlorination (Könemann & Van Leeuwen, 1980; Oliver & Niimi, 1983; Sabljic, 1987; Koelmans & Jimenez, 1994; Wang et al., 1997) and with increasing temperature (Koelmans & Jimenez, 1994).

Mean bioconcentration factors (BCFs) (dry weight) for phytoplankton increased from 4700 for 1,2,3-TCB at 4.5 °C to 26 000 for PeCB at 38.6 °C (Koelmans & Jimenez, 1994). Wang et al. (1997) found significant differences in the accumulation of chlorobenzenes by different marine algal species, with BCFs (dry weight) ranging from 600 to 3000 for 1,2,3,4-TeCB and from 1000 to 6000 for PeCB.

BCFs ranging from 270 for 1,2-DCB to 20 000 for PeCB were reported for laboratory studies on rainbow trout (*Oncorhynchus mykiss*) (Oliver & Niimi, 1983). BCFs for a variety of fish species ranged from 7000 to 24 000 (lipid weight) for 1,2,4-TCB, with a positive correlation between bioaccumulation and lipid content (Geyer et al., 1985). Galassi & Calamari (1983) found BCFs (lipid weight) ranging from 4000 to 22 000 for 1,2,3- and 1,2,4-TCB in rainbow trout, with newly hatched fish accumulating 2–4 times the amount found in eyed eggs or young fish (alevins). Qiao et al. (2000) report that gill uptake of 1,2,4-TCB and PeCB could account for 98% of the body burden. Uptake of trichlorobenzenes, tetrachlorobenzenes, and PeCB was significantly reduced by the presence of suspended particles (Schrap & Opperhuizen, 1990). However, PeCB was found to be readily desorbed from sediments with a low organic carbon content and subsequently accumulated by fish via the gills (Qiao & Farrell, 1996). The rate of elimination of chlorobenzenes decreases with increasing chlorination (Melancon & Lech, 1985; De Boer et al., 1994). Elimination half-lives for dichlorobenzenes to PeCB in laboratory-exposed fish ranged from 0.05 to 1.6 days (Melancon & Lech, 1985). However, for eels (*Anguilla anguilla*) transferred from a contaminated lake to a "clean" lake, elimination half-lives of >300 days were reported for tetrachlorobenzenes and PeCB (De Boer et al., 1994). Sijm & Van der Linde (1995) calculated elimination rate constants and predicted elimination half-lives for 1,2,3-TCB to be 40 days in small fish, such as guppies (*Poecilia reticulata*), and >5 years in larger and/or fatty fish.

The coefficient of adsorption onto sediment influences the uptake into terrestrial plants and sediment-living aquatic invertebrates; the degree of chlorination is also correlated with uptake (Knezovich & Harrison, 1988; IPCS, 1991a). Under non-equilibrium conditions, BCFs for chironomid midge larvae exposed to sediment-bound chlorobenzenes were 5, 29, and 225 for MCB, 1,2-DCB, and 1,2,4-TCB, respectively. BCFs were best correlated with the concentrations of the chlorobenzenes in the interstitial water (Knezovich & Harrison, 1988).

The tri- and tetrachlorinated benzenes may be taken up by plants, as indicated by the root concentration factor of 19 litres/kg reported for 1,2,4-TCB (Dietz & Schnoor, 2001).

However, the prediction of BCFs is more difficult for terrestrial plants than for aquatic organisms because of the complex nature of the root soil interface combined with gaseous uptake by aerial parts (Scheunert et al., 1994). Topp et al. (1986) compared the uptake of chlorobenzenes by plants from the soil and via the air in closed, aerated laboratory systems. A negative correlation was demonstrated between the BCF and the soil adsorption coefficient (based on soil organic matter content) for the uptake into the roots of barley. The adsorption of chlorobenzenes onto soil organic matter increased with increasing chlorination. However, expression of uptake in barley roots in relation to the soil interstitial water concentration of the chlorobenzenes produced a positive correlation between the BCF and the octanol/water partition coefficients. Higher chlorinated chlorobenzenes, therefore, are most readily taken up by the plant roots when they are available in soil interstitial water. This will occur particularly in sandy soils with low organic matter content. In a later study, Topp et al. (1989) found that after growth in soil containing 2 µg each of 1,2,4-TCB and PeCB per kg dry weight, harvested barley grain contained 73 and 82 µg/plant, respectively. The concentrations in the dry grain were 0.05 and 0.06 mg/kg for 1,2,4-TCB and PeCB, respectively. In further studies on soybeans (*Glycine max*), linear correlations were found between equilibrium tissue/water coefficients, the octanol/water partition coefficient, and measured lipid content (Tam et al., 1996). The bioconcentration of chlorobenzenes into excised soybean (*Glycine max*) roots increased exponentially with increasing octanol/water partition coefficient (Kraaij & Connell, 1997). Wang & Jones (1994b) concluded that the total amount of chlorobenzenes taken up by carrots grown in sewage sludge-amended and spiked soils was low (<1%) compared with other loss pathways from the soil, principally volatilization.

Belfroid et al. (1994) calculated BCFs for earthworms (*Eisenia andrei*) of 104 and 156 for 1,2,3,4-TeCB and PeCB in soil; BCFs based on interstitial water were 67 000 and 307 000, respectively, and were found to be similar to BCFs found for worms exposed in water alone (Belfroid et al., 1993). BCFs for earthworms exposed via water show a clear increase in uptake of chlorobenzenes with increasing chlorination, and steady-state concentrations are reached within 5 days (Belfroid et al., 1993). Elimination rate constants reveal that chlorobenzene loss decreases with increasing chlorination. A monophasic elimination curve was observed in water, whereas biphasic elimination was found in the presence of soil (Belfroid et al., 1993); elimination rates in soil experiments were significantly increased by the addition of organic matter (Belfroid & Sijm, 1998). Feeding studies have revealed that earthworms can also take up chlorobenzenes via food. In studies with field-contaminated soil, steady-state concentrations in worms were much lower than in laboratory studies, suggesting decreased bioavailability of chlorobenzenes (Belfroid et al., 1995).

6. ENVIRONMENTAL LEVELS

Chlorobenzene (MCB, dichlorobenzenes, and trichlorobenzenes) concentrations have previously been reported in ambient air, with mean concentrations in the order of 0.1 $\mu\text{g}/\text{m}^3$ and maximum levels of up to 100 $\mu\text{g}/\text{m}^3$ at hazardous waste sites (IPCS, 1991a). Popp et al. (2000) measured tetrachlorobenzenes and PeCB in air sampled from two industrially contaminated sites and a reference site in Germany in 1998. Mean gas-phase concentrations of tetrachlorobenzenes and PeCB at the contaminated sites ranged from 5.7 to 30.9 pg/m^3 and from 10.2 to 28 pg/m^3 , respectively. Mean concentrations at the control site ranged from 6.4 to 10.6 pg/m^3 . Particulate-bound chlorobenzenes accounted for 1.9% of the total concentrations. A low proportion of particulate-bound chlorobenzenes was also reported in air sampled from the Bering and Chukchi seas in 1993 (Strachan et al., 2001). Mean gas-phase concentrations for the Bering Sea were 1.1, 4.0, and 6.6 pg/m^3 for 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB, respectively, and for the Chukchi Sea, 2.8, 10, and 14 pg/m^3 , respectively. Mean chlorobenzene concentrations at four sites throughout Michigan, USA (1992–1994), ranged from 22 to 30 pg/m^3 for 1,2,4,5-TeCB, from 40 to 53 pg/m^3 for 1,2,3,4-TeCB, and from 35 to 69 pg/m^3 for PeCB (Hermanson et al., 1997). Annual mean concentrations for southern Ontario, Canada (1988–1989), were $>5.3 \text{ pg}/\text{m}^3$ for 1,2,3,4-TeCB and $>8.0 \text{ pg}/\text{m}^3$ for PeCB (Hoff et al., 1992). Higher concentrations have been reported in close proximity to pollution sources. A concentration of 5 $\mu\text{g}/\text{m}^3$ for tri- and tetrachlorobenzenes was found within 200 m of an electro-industrial plant in Slovenia (Jan et al., 1994). Seasonal variations in the concentrations of 1,4-DCB in ambient air have also been reported, with concentrations increasing with increasing temperature (Hanai et al., 1985).

Chlorobenzenes have also been detected in rainwater, their presence presumably being due to transfer from the ambient air. Concentrations of all three dichlorobenzene isomers and 1,2,4-TCB in rainwater were less than 10 ng/litre at selected sites in Oregon and California, USA (Pankow et al., 1983). In the United Kingdom, 1,4-DCB was detected in rainwater at a mean concentration of $10 \pm 5 \text{ ng}/\text{litre}$ (Fielding et al., 1981).

In 12 sewage sludges in the United Kingdom, the concentrations of chlorobenzenes ranged from $<0.01 \text{ mg}/\text{kg}$ dry weight for PeCB to 40.2 mg/kg dry weight for 1,3-DCB, with a general reduction in concentration with increased chlorine substitution (Rogers et al., 1989). Further sampling of United Kingdom sewage sludges revealed chlorobenzene concentrations ranging from 35 100 to 192 000 mg/kg dry weight for MCB, from 13 to 4110 mg/kg for dichlorobenzenes, from 2 to 1070 mg/kg for trichlorobenzenes, from 0.2 to 101 mg/kg for tetrachlorobenzenes, and from 2 to 37 mg/kg for PeCB (Wang & Jones, 1994c). Analysis of archived sludge samples showed that concentrations of 1,4-DCB increased over the period 1942–1961, whereas other chlorobenzenes increased in concentration only from 1954 onwards (Wang et al., 1992).

Data on levels of the lower chlorinated benzenes (MCB, dichlorobenzenes, and trichlorobenzenes) in wastewater indicate that MCB is detected the most often and at the highest concentrations, occasionally exceeding 1 mg/litre . Chlorobenzene concentrations in US wastewater have been reported to range from 11 to 6400 $\mu\text{g}/\text{litre}$ for MCB, from 10 to 860 $\mu\text{g}/\text{litre}$ for dichlorobenzenes, and from 12 to 607 $\mu\text{g}/\text{litre}$ for trichlorobenzenes (IPCS, 1991a).

Concentrations of chlorobenzenes in surface waters are generally in the ng/litre to $\mu\text{g}/\text{litre}$ range, with maximum concentrations up to 0.2 mg/litre in areas close to industrial sources (IPCS, 1991a). Mean concentrations of dissolved chlorobenzenes in the Bering and Chukchi seas ranged from 3 to 10 pg/litre for 1,2,3-TCB, from 15 to 36 pg/litre for 1,2,3,4-TeCB, and from 9 to 36 pg/litre for PeCB (Strachan et al., 2001). Higher chlorobenzene levels have been detected in coastal waters and estuaries, with Dutch coastal waters containing mean concentrations ranging from 9 to 117 ng/litre for dichlorobenzenes and from 0.7 to 1.6 ng/litre for trichlorobenzenes (Van de Meent et al., 1986) and Japanese coastal waters containing mean dissolved concentrations ranging from 24.3 ng/litre for 1,3-DCB to 0.25 ng/litre for tetrachlorobenzenes (Masunaga et al., 1996b). Waters of the Scheldt estuary (The Netherlands) contained chlorobenzene concentrations ranging from <130 to 315 ng/litre for dichlorobenzenes, from <25 to 320 ng/litre for trichlorobenzenes, and from <45 to 135 ng/litre for tetrachlorobenzenes (Van Zoest & Van Eck, 1991); more recent sampling revealed MCB concentrations ranging from 5 to 31.5 ng/litre (Huybrechts et al., 2000). Chlorobenzene concentrations of up to 500 ng/litre have been reported for MCB in the Tees Estuary, United Kingdom (Law et al., 1991), and for 1,3-DCB in Yokkaichi Port, Ise Bay, Japan, during 1988 (Masunaga et al., 1991a). Mean chlorobenzene concentrations in the Forth Estuary, United Kingdom, during 1987 ranged from <0.1 to 790 ng/litre for dichlorobenzenes, from 4 to 5500 ng/litre for trichlorobenzenes, from <0.04 to 20 ng/litre for tetrachlorobenzenes, and from <0.01 to 40 ng/litre for PeCB. The predominant isomers detected were 1,2,3- and 1,2,4-TCB, and these were found near industrial effluent discharges (Rogers et al., 1989; Harper et al., 1992). Further studies in 1990 revealed 1,2,3- and 1,2,4-TCB concentrations ranging up to 51 and 84 ng/litre , respectively (Harper et al., 1992).

The highest chlorobenzene concentrations in surface waters have been reported for river waters in heavily populated and/or industrialized areas. Mean concentrations in the river Besos, Spain, were 260 ng/litre for MCB, 600 ng/litre for 1,4-DCB, 5000 ng/litre for 1,2-DCB and 1,3-DCB, 1100 ng/litre for 1,2,3-TCB, and 8100 ng/litre for 1,2,4-TCB (Gomez Belinchon et al., 1991). Concentrations of MCB and 1,4-DCB ranging from non-detected to $>10 \mu\text{g}/\text{litre}$ have been reported for both compounds in water from the Ohio River (US EPA, 1985). Elder et al. (1981) reported trichlorobenzene concentrations (isomer not specified) ranging from 0.1 to 8 $\mu\text{g}/\text{litre}$ in water from Niagara Falls, New York, USA. Corresponding concentrations of tetrachlorobenzene ranged from 0.1 to 200 $\mu\text{g}/\text{litre}$. Concentrations of PeCB in water sampled from the Great Lakes ranged from not detected to 0.0006 $\mu\text{g}/\text{litre}$ (Oliver & Nicol, 1982). Concentrations in water sampled from the rivers and estuary of Osaka (a major urban area of Japan) ranged from 0.2 to 30 $\mu\text{g}/\text{litre}$ for MCB, from 0.17 to 130 $\mu\text{g}/\text{litre}$ for 1,4-DCB, from 0.2 to 10 $\mu\text{g}/\text{litre}$ for 1,2-DCB, from 0.16 to 0.35 $\mu\text{g}/\text{litre}$ for 1,2,4-TCB, and from 0.18 to 0.30 $\mu\text{g}/\text{litre}$ for 1,2,3-TCB (Yamamoto et al., 1997).

Mean chlorobenzene concentrations in sediment from the Bering and Chukchi seas ranged from 0.02 to 0.41 $\mu\text{g}/\text{kg}$ for 1,2,3-TCB, from 0.08 to 0.87 $\mu\text{g}/\text{kg}$ for 1,2,3,4-TeCB, and from 0.33 to 0.4 $\mu\text{g}/\text{kg}$ for PeCB (Strachan et al., 2001). Mean concentrations in coastal sediments from Ise Bay, Japan, were 4.8 $\mu\text{g}/\text{kg}$ for 1,2,4-TCB, 2.3 $\mu\text{g}/\text{kg}$ for 1,2-DCB, 1.9 $\mu\text{g}/\text{kg}$ for 1,3-DCB, and $<0.15 \mu\text{g}/\text{kg}$ for 1,3,5-TCB, tetrachlorobenzenes, and PeCB (Masunaga et al., 1991b). Lee & Fang (1997) reported mean values for the Tsen-wen estuary, Taiwan, of 3.2 $\mu\text{g}/\text{kg}$ for 1,2-DCB, 20.7 $\mu\text{g}/\text{kg}$ for 1,3-DCB, and 11.2 $\mu\text{g}/\text{kg}$ for 1,2,4-TCB.

Lake Garda (Italy) contained mean sediment PeCB concentrations of 0.2 $\mu\text{g}/\text{kg}$ dry weight (Bossi et al., 1992), whereas Lake Superior (Hamilton Harbour, Canada) contained levels ranging from 3.6 $\mu\text{g}/\text{kg}$ for PeCB to 80 $\mu\text{g}/\text{kg}$ for 1,4-DCB (Onuska & Terry, 1985). Sediment samples from the river Elbe,

Germany, ranged from 30 to 740 µg/kg dry weight for MCB, from 20 to 1060 µg/kg for dichlorobenzenes (1,2- and 1,4-DCB), from 1 to 115 µg/kg for trichlorobenzenes (1,2,3- and 1,2,4-TCB), from 1 to 27 µg/kg for tetrachlorobenzenes, and from 1 to 14 µg/kg for PeCB (Götz et al., 1993), whereas samples from the river Rhine contained concentrations ranging from 40 to 240 µg/kg dry weight for dichlorobenzenes, from <10 to 20 µg/kg for trichlorobenzenes, and from <0.5 to 2 µg/kg for PeCB (Alberti, 1983).

Chlorobenzene levels in uncontaminated soils are generally less than 0.4 mg/kg for dichlorobenzene congeners and less than 0.1 mg/kg for other chlorobenzene congeners (Wang et al., 1995). Multiple applications of sewage sludge can increase the chlorobenzene content in sludge-amended soil compared with control soils. However, Wang et al. (1995) found that most chlorobenzenes disappear rapidly on cessation of sludge application, with around 10% remaining 30 years later. They found that 1,4-DCB levels increased significantly in United Kingdom soils during the 1960s to a maximum mean value in 1967 of 10 mg/kg in control soils and 16.6 mg/kg in sludge-amended soils. Analysis of subsoil from a former pesticide factory in Germany showed that tetrachlorobenzenes and PeCB were dominant in the upper soil layers (up to 1.9 m), accounting for 80% of chlorobenzenes, with 1,2,3,4-TeCB and PeCB accounting for 44% and 24%, respectively. At depths between 1.9 and 5.5 m, trichlorobenzenes were more dominant, accounting for 60%, with 1,2,4-TCB accounting for 37% (Feidieker et al., 1994). Total chlorobenzene concentrations ranged from 1.5 to 18 400 mg/kg.

Mean chlorobenzene concentrations in bivalves from US coastal waters ranged from <0.25 to 28.2 µg/kg dry weight for 1,2,4,5-TeCB, from <0.25 to 10 µg/kg for 1,2,3,4-TeCB, and from <0.25 to 13.3 µg/kg for PeCB (Wade et al., 1998). Aquatic insects from a variety of Canadian sites contained mean PeCB concentrations ranging from <0.49 to 21.4 µg/kg dry weight (Ciborowski & Corkum, 1988). Concentrations in freshwater and marine fish from contaminated areas range from 0.1 to 50 µg/kg wet weight, with higher chlorinated compounds generally present at the highest concentrations (IPCS, 1991a). The eggs of fish-eating birds contained mean PeCB levels of 1.2 and 4.4 µg/kg from two sites in Puget Sound, USA (Cobb et al., 1994). Waterfowl from Lake Ontario, Canada, contained mean chlorobenzene concentrations ranging from 0.3 to 1.7 µg/kg wet weight for 1,2,3,4-TeCB and from 0.65 to 33.4 µg/kg for 1,2,4,5-TeCB (Gebauer & Weseloh, 1993). Mean concentrations in Arctic marine mammal blubber ranged from 1 to 9.7 µg/kg wet weight for 1,2,3,4-TeCB and from 16.8 to 20.2 µg/kg for PeCB (Muir et al., 1992; Weis & Muir, 1997).

7. EFFECTS ON ORGANISMS IN THE LABORATORY AND FIELD

7.1 Aquatic environment

The acute toxicity of chlorobenzenes to aquatic organisms is presented in Table 3. Forty-eight-hour EC₅₀s for diatoms range from 8 to 235 000 µg/litre. For freshwater invertebrates, 48-h EC₅₀s range from 10 µg/litre for PeCB to >530 000 µg/litre for 1,2,4,5-TeCB. Ninety-six-hour LC₅₀s for fish range from 135 for PeCB in the freshwater guppy (*Poecilia reticulata*) to 21 000 µg/litre for 1,2,4-TCB in the saltwater sheepshead minnow (*Cyprinodon variegatus*).

Table 3: Acute toxicity of chlorobenzenes to aquatic species.

Organism	End-point	Chlorobenzene	Test conditions ^a	Concentration (µg/litre)	Reference
Microorganisms — Saltwater					
Diatom (<i>Cyclotella meneghiniana</i>)	48-h EC ₅₀ (DNA measurement)	MCB	M	235 740	Figueroa & Simmons (1991)
		1,2-DCB	M	23 330	Figueroa & Simmons (1991)
		1,3-DCB	M	51 880	Figueroa & Simmons (1991)
		1,4-DCB	M	34 300	Figueroa & Simmons (1991)
		1,2,3-TCB	M	6420	Figueroa & Simmons (1991)
		1,2,4-TCB	M	2830	Figueroa & Simmons (1991)
		1,3,5-TCB	M	590	Figueroa & Simmons (1991)
		1,2,3,5-TeCB	M	1370	Figueroa & Simmons (1991)
		1,2,3,4-TeCB	M	1390	Figueroa & Simmons (1991)
		1,2,4,5-TeCB	M	270	Figueroa & Simmons (1991)
		PeCB	M	8	Figueroa & Simmons (1991)
Invertebrates — Freshwater					
Water flea (<i>Daphnia magna</i>)	48-h EC ₅₀ /LC ₅₀	MCB	C M	585.52	Rose et al. (1998)
		MCB	C N	12 900–17 300 ^b	Cowgill et al. (1985)
		MCB	C N	5810	Abernethy et al. (1986)

		MCB	C N	86 000	LeBlanc (1980)
		1,2-DCB	C N	2352	Abernethy et al. (1986)
		1,2-DCB	N	740–2200 ^c	Canton et al. (1985)
		1,2-DCB	C M	4200–7400	Richter et al. (1983)
		1,2-DCB	C N	2400	LeBlanc (1980)
		1,3-DCB	N	1200–6800 ^c	Canton et al. (1985)
		1,3-DCB	N	10 500–13 500	Gersich et al. (1986)
		1,3-DCB	C N	28 000	LeBlanc (1980)
		1,4-DCB	N	700–2200 ^c	Canton et al. (1985)
		1,4-DCB	C N	11 000	LeBlanc (1980)
		1,2,3-TCB	C N	1452	Abernethy et al. (1986)
		1,2,4-TCB	C M	1700–2100	Richter et al. (1983)
		1,2,4-TCB	C N	50 000	LeBlanc (1980)
		1,2,4,5-TeCB	C N	>530 000	LeBlanc (1980)
		PeCB	C N	300	Abernethy et al. (1986)
Water flea (<i>Ceriodaphnia dubia</i>)	48-h EC ₅₀ /LC ₅₀	MCB	M C	7900–47 000	Rose et al. (1998)
		MCB	C N	8900–11 100 ^d	Cowgill et al. (1985)
		1,2-DCB	M C	661.5	Rose et al. (1998)
		1,2,4-TCB	M C	308	Rose et al. (1998)
		1,2,3,4-TeCB	M C	130	Rose et al. (1998)
		PeCB	M C	10	Rose et al. (1998)
Midge (<i>Tanytarsus dissimilis</i>)	48-h LC ₅₀	1,2-DCB		2300–11 800	Call et al. (1979)
		1,4-DCB		13 000	Call et al. (1983)
Midge (<i>Chironomus thummi</i>)	48-h LC ₅₀	1,4-DCB	C M	1200	Roghair et al. (1994)
		1,2,3-TCB	C M	1700	Roghair et al. (1994)
		1,2,3,4-TeCB	C M	540–730	Roghair et al. (1994)
		PeCB	C M	230–320	Roghair et al. (1994)
Invertebrates — Saltwater					
Fleshy prawn (<i>Penaeus chinensis</i>)	96-h LC ₅₀	MCB		1720	Yin & Lu (1993)
Crab (<i>Portunus pelagicus</i>)	96-h EC ₅₀ (growth)	MCB	C N	748	Mortimer & Connell (1994)
		1,4-DCB	C N	201	Mortimer & Connell (1994)
		1,2,3-TCB	C N	173	Mortimer & Connell (1994)
		1,2,3,4-TeCB	C N	410	Mortimer & Connell (1994)
		PeCB	C N	87	Mortimer & Connell (1994)
Grass shrimp (<i>Palaemonetes pugio</i>)	96-h LC ₅₀	1,2-DCB	M	9400	Curtis et al. (1979)
		1,4-DCB	M	69 000	Curtis et al. (1979)
		1,2,4-TCB	M	540	Clark et al. (1987)
Opossum shrimp (<i>Americamysis</i>)	96-h LC ₅₀	1,3-DCB		2850	US EPA (1978)

bahia)		1,2,4-TCB		450	US EPA (1978)
		1,2,4,5-TeCB		1480	US EPA (1978)
		PeCB		160	US EPA (1978)
Fish — Freshwater					
Rainbow trout (<i>Oncorhynchus mykiss</i>)	96-h LC ₅₀	MCB	M	4700	Dalich et al. (1982)
		1,2-DCB		1520–1580	Call et al. (1979)
		1,4-DCB		880	Mayer & Ellersieck (1986)
		1,4-DCB		1120	Call et al. (1983)
		1,2,4-TCB		1530	Call et al. (1983)
		1,2,4,5-TeCB	N	1200–10 000 ^e	Van Leeuwen et al. (1985)
		PeCB		190	Call et al. (1979)
	16-day LC ₅₀	MCB	C N	90	Birge et al. (1979a)
14-day LC ₅₀	1,4-DCB	C M	800	Calamari et al. (1983)	
Fathead minnow (<i>Pimephales promelas</i>)	96-h LC ₅₀	MCB	C M	7700	Marchini et al. (1993)
		MCB	M	16 900	Geiger et al. (1990)
		1,2-DCB	M	6027	Sijm et al. (1993)
		1,3-DCB	M	7800	Carlson & Kosian (1987)
		1,4-DCB	M	4200	Carlson & Kosian (1987)
		1,2,4-TCB	M	2760	Carlson & Kosian (1987)
		1,2,4-TCB	M	2990	Geiger et al. (1990)
		1,2,3,4-TeCB	M	1100	Carlson & Kosian (1987)
		1,2,4,5-TeCB		89–460	Brooke (1991)
Goldfish (<i>Carassius auratus</i>)	96-h LC ₅₀	MCB	C N	2370–3480 ^f	Birge et al. (1979a)
Guppy (<i>Poecilia reticulata</i>)	96-h LC ₅₀	1,2,3-TCB	M	348	Van Hoogen & Opperhuizen (1988)
		1,2,3-TCB	M	365	Van Hoogen & Opperhuizen (1988)
		PeCB	M	135	Van Hoogen & Opperhuizen (1988)
	14-day LC ₅₀	MCB	C N	24 964	Könemann (1981)
		1,2-DCB	C N	5852	Könemann (1981)
		1,3-DCB	C N	7367	Könemann (1981)
		1,4-DCB	C N	3957	Könemann (1981)
		1,2,4-TCB	C N	2393	Könemann (1981)
		1,3,5-TCB	C N	3302	Könemann (1981)
	14-day LC ₅₀	1,2,4,5-TeCB	C N	305	Könemann (1981)
		PeCB	C N	177	Könemann (1981)
Mosquitofish (<i>Gambusia affinis</i>)	96-h LC ₅₀	1,2,3-TCB	C M	2196	Chaisuksant et al. (1998)
	96-h LC ₅₀	PeCB	C M	200	Chaisuksant et al. (1998)

Largemouth bass (<i>Micropterus salmoides</i>)	7.5-day LC ₅₀	MCB	C M	50–60 ^g	Birge et al. (1979b)
Zebrafish (<i>Brachydanio rerio</i>)	28-day LC ₅₀	MCB	M	10 300	Van Leeuwen et al. (1990)
	7- to 28-day NOEC	MCB	M	8500	Van Leeuwen et al. (1990)
	14- to 28-day NOEC	1,4-DCB	M	2100	Van Leeuwen et al. (1990)
	28-day LC ₅₀	1,2,3-TCB	M	990	Van Leeuwen et al. (1990)
	14- to 28-day NOEC	1,2,3-TCB	M	450	Van Leeuwen et al. (1990)
	14- to 28-day NOEC	1,2,4-TCB	M	450	Van Leeuwen et al. (1990)
	28-day LC ₅₀	1,2,3,4-TeCB	M	410	Van Leeuwen et al. (1990)
	7- to 21-day NOEC	1,2,3,4-TeCB	M	310	Van Leeuwen et al. (1990)
	7- to 28-day NOEC	PeCB	M	110	Van Leeuwen et al. (1990)
Fish — Saltwater					
Dover sole (<i>Solea solea</i>)	96-h LC ₅₀	MCB	C M	5821	Furay & Smith (1995)
European flounder (<i>Platichthys flesus</i>)	96-h LC ₅₀	MCB	C M	6609	Furay & Smith (1995)
		1,2-DCB	C M	4616	Furay & Smith (1995)
		1,2,4-TCB	C M	8585	Furay & Smith (1995)
Sheepshead minnow (<i>Cyprinodon variegatus</i>)	96-h LC ₅₀	MCB	N	10 000	Heitmuller et al. (1981)
		1,2-DCB	N	9700	Heitmuller et al. (1981)
		1,3-DCB	N	7800	Heitmuller et al. (1981)
		1,4-DCB	N	7400	Heitmuller et al. (1981)
		1,2,4-TCB	N	21 000	Heitmuller et al. (1981)
		1,2,3,5-TeCB	N	3700	Heitmuller et al. (1981)
		1,2,4,5-TeCB	N	800	Heitmuller et al. (1981)
		1,2,4,5-TeCB	M	330	Ward et al. (1981)
		PeCB	N	800	Heitmuller et al. (1981)
Sheepshead minnow (<i>Cyprinodon variegatus</i>)	28-day NOEC (growth)	PeCB		18–86 ^h	Hansen & Cripe (1991)
	28-day NOEC (survival)	PeCB		19–120 ^h	Hansen & Cripe (1991)

a C = test carried out in a closed exposure system; M = measured exposure concentration; N = nominal exposure concentration.

b Range of values for differences in temperature: 12 900 refers to 20 °C, 17 300 refers to 24 °C.

c Range indicated difference between EC₅₀ (lower value) and LC₅₀ (upper value).

d Range of values for differences in temperature: 8900 refers to 20 °C, 11 100 refers to 24 °C.

e Range of values for different life stages: 10 000 refers to all egg stages and sac fry; 1200 refers to early fry.

f Range of values for differences in water hardness: 2370 refers to 200 mg calcium carbonate/litre; 3480 refers to 50 mg calcium carbonate/litre.

g Range of values for differences in water hardness: 50 refers to 50 mg calcium carbonate/litre; 60 refers to 200 mg calcium carbonate/litre.

h Range of values due to interlaboratory comparisons.

Chronic toxicity of chlorobenzenes to aquatic organisms is presented in Table 4. Seventy-two-hour EC_{50} s for green algae range from 5280 $\mu\text{g/litre}$ for 1,3-DCB to 200 000 $\mu\text{g/litre}$ for MCB. Chronic NOECs for freshwater invertebrates range from 32 $\mu\text{g/litre}$ (PeCB) to 19 000 $\mu\text{g/litre}$ (MCB) for reproduction and from <1400 to 3890 $\mu\text{g/litre}$ (MCB) for survival. In fish, NOECs range from 18 $\mu\text{g/litre}$ for PeCB to 8500 $\mu\text{g/litre}$ for MCB. The data in this table include the standardized ecotoxicological tests, although any species shown to be more sensitive to the effects of chlorobenzenes have also been included. Care must be taken when interpreting these data, as chlorobenzenes may volatilize from the test system (with the lower chlorinated compounds generally more volatile), thus reducing actual exposure.

Table 4: Chronic toxicity of chlorobenzenes to aquatic organisms.

Organism	End-point	Chlorobenzene	Test conditions ^a	Concentration ($\mu\text{g/litre}$)	Reference
Microorganisms — Freshwater					
Green algae (<i>Pseudokirchneriella subcapitata</i>) ^b	72-h EC_{50} (population)	MCB	N	202 000	US EPA (1978)
		1,2-DCB	N	76 100	US EPA (1978)
		1,3-DCB	N	5280	US EPA (1978)
		1,4-DCB	N	77 500	US EPA (1978)
		1,2,4-TCB	N	21 700	US EPA (1978)
		1,2,3,5-TeCB	N	14 700	US EPA (1978)
		PeCB	N	13 000	US EPA (1978)
Invertebrates — Freshwater					
Water flea (<i>Daphnia magna</i>)	10-day LC_{50}	MCB	C N	16 000	Cowgill & Milazzo (1991)
	9- to 11-day NOEC (survival)	MCB	C N	<1400	Cowgill & Milazzo (1991)
	9- to 11-day EC_{50} (reproduction)	MCB	C N	15 000–19 000 ^c	Cowgill & Milazzo (1991)
	9- to 11-day NOEC (reproduction)	MCB	C N	6500–11 000 ^d	Cowgill & Milazzo (1991)
	Four-brood EC_{50} (reproduction)	MCB	M	1912	De Wolf et al. (1988)
	Four-brood NOEC (reproduction)	MCB	M	1004	De Wolf et al. (1988)
	14-day EC_{50} (reproduction)	1,2-DCB	C M	550	Calamari et al. (1983)
	16-day EC_{50} (reproduction)	1,3-DCB	M	1400	Hermens et al. (1984)
	14-day EC_{50} (reproduction)	1,4-DCB	C M	930	Calamari et al. (1983)
	14-day EC_{50} (reproduction)	1,2,3-TCB	C M	200	Calamari et al. (1983)
	Four-brood EC_{50} (reproduction)	1,2,4-TCB	M	330	De Wolf et al. (1988)
	Four-brood NOEC (reproduction)	1,2,4-TCB	M	182	De Wolf et al. (1988)
	14-day EC_{50} (reproduction)	1,2,4-TCB	C M	450	Calamari et al. (1983)
	Four-brood EC_{50} (reproduction)	1,2,3,4-TeCB	M	90	De Wolf et al. (1988)
	Four-brood NOEC (reproduction)	1,2,3,4-TeCB	M	55	De Wolf et al. (1988)
	Four-brood EC_{50} (reproduction)	PeCB	M	61	De Wolf et al. (1988)
Four-brood NOEC (reproduction)	PeCB	M	32	De Wolf et al. (1988)	

Water flea (<i>Ceriodaphnia dubia</i>)	7-day LC ₅₀	MCB	C N	24 000	Cowgill & Milazzo (1991)
	7- to 10-day NOEC (survival)	MCB	C N	3890	Cowgill & Milazzo (1991)
	7- to 10-day EC ₅₀ (reproduction)	MCB	C N	14 000–26 000 ^c	Cowgill & Milazzo (1991)
	7- to 10-day NOEC (reproduction)	MCB	C N	12 000–19 000 ^f	Cowgill & Milazzo (1991)
Invertebrates — Saltwater					
Crab (<i>Portunus pelagicus</i>)	40-day NOEC (growth)	MCB	C N	125	Mortimer & Connell (1995)
		1,4-DCB	C N	31	Mortimer & Connell (1995)
		1,2,3-TCB	C N	25	Mortimer & Connell (1995)
		1,2,3,4-TeCB	C N	17	Mortimer & Connell (1995)
		PeCB	C N	5	Mortimer & Connell (1995)
Fish — Freshwater					
Rainbow trout (<i>Oncorhynchus mykiss</i>)	85-day LOEC (growth)	1,2,4-TCB	M	36.3–90.75	Hodson et al. (1991)
	85-day LOEC (survival)	1,2,4-TCB	M	454–853	Hodson et al. (1991)
Fish — Saltwater					
Sheepshead minnow (<i>Cyprinodon variegatus</i>)	28-day NOEC (growth)	PeCB		18–86 ^g	Hansen & Cripe (1991)
	28-day NOEC (survival)	PeCB		19–120 ^g	Hansen & Cripe (1991)

a C = test carried out in a closed exposure system; M = measured exposure concentration; N = nominal exposure concentration.

b Previously known as *Selenastrum capricornutum*.

c Range of EC₅₀ values refers to different measurements of reproduction. EC₅₀ values for progeny, broods, and mean brood size were 15 000, 19 000, and 16 000 µg/litre, respectively.

d Range of NOEC values refers to different measurements of reproduction. NOEC values for progeny, broods, and mean brood size were 11 000, 11 000, and 6500 µg/litre, respectively.

e Range of EC₅₀ values refers to different measurements of reproduction. EC₅₀ values for progeny, broods, and mean brood size were 14 000, 26 000, and 22 000 µg/litre, respectively.

f Range of NOEC values refers to different measurements of reproduction. NOEC values for progeny, broods, and mean brood size were 19 000, 19 000, and 12 000 µg/litre, respectively.

g Range of values due to interlaboratory comparisons.

Concentrations of MCB greater than 2.5 mmol/litre (282 mg/litre) were found to be toxic to *Pseudomonas* sp. strain RHO1 cells. Cells that had previously been exposed to MCB demonstrated toxicity at concentrations greater than 3.5 mmol/litre (394 mg/litre) (Fritz et al., 1992). Figueroa & Simmons (1991) measured the effect of chlorobenzenes on diatom (*Cyclotella meneghiniana*) DNA. DNA quantification was used as a biological indicator of cellular biomass. Toxicity increased with increasing chlorination and was related to physicochemical and structural properties. Toxicity was explained in terms of partitioning between the aqueous and biological phases.

The acute toxicity of MCB to daphnids (*Daphnia magna* and *Ceriodaphnia dubia*) was higher at 20 °C than at 24 °C. Forty-eight-hour LC₅₀ values for *Daphnia magna* and *Ceriodaphnia dubia* were 12 900 and 8900 µg/litre at 20 °C and 17 300 and 11 100 µg/litre at 24 °C, respectively (Cowgill et al., 1985). Rose et al. (1998) reported that *Ceriodaphnia dubia* was approximately 4 times more sensitive to chlorobenzenes than *Daphnia magna*. Richter et al. (1983) compared the acute 48-h LC₅₀ values and chronic 28-day NOEC values of 1,3-DCB and 1,2,4-TCB for *Daphnia magna*. The ratio of acute:chronic toxicity in *Daphnia* was calculated to be 5.1 and 3.0 for 1,3-DCB and 1,2,4-TCB, respectively. Both the number of young produced per adult and the length of adults as expressions of chronic toxicity were equally sensitive for determining statistically significant effects (Richter et al., 1983; De Wolf et al., 1988). The toxicity of chlorobenzenes to midge (*Chironomus riparius*) larvae increased with increasing chlorination. Ratios of LC₅₀:NOEC values were 13, 5.0, 2.3–6.6, and 2.4 for 1,4-DCB, 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB, respectively (Roghair et al., 1994). The toxicity of chlorobenzenes to sand crab (*Portunus pelagicus*) increased during the moulting stage. Toxicity also increased with increasing chlorination and octanol/water partition coefficient (Mortimer & Connell, 1994).

The NOEC, with reference to effects on survival and embryo hatchability for zebrafish (*Brachydanio rerio*) eggs exposed to MCB, 1,4-DCB, 1,2,3-TCB, 1,2,3,4-TeCB, or PeCB, was reported to be the same, regardless of whether the exposure period was 7, 14, 21, or 28 days (Van Leeuwen et al., 1985). As the test solutions were replaced 3 times per week, the identical NOECs are unlikely to be due to evaporation of the chlorobenzenes. The effect of dissolved

oxygen on the toxicity of 1,2,4-TCB to fathead minnows (*Pimephales promelas*) was examined by Carlson (1987). Fish were exposed to 1,2,4-TCB concentrations of up to 920 µg/litre at 4.5, 5.6, or 8.1 mg dissolved oxygen per litre. At all dissolved oxygen concentrations, no effects on survival or growth were reported at 1,2,4-TCB concentrations up to 500 µg/litre. Reduced survival and mean body weight were reported in fish exposed to 920 µg/litre, with toxicity increased in fish from the low dissolved oxygen group. Van Hoogen & Opperhuizen (1988) calculated the chlorobenzene concentration in guppy (*Poecilia reticulata*) tissues following lethality in acute toxicity experiments. Lethal concentrations were the same for 1,2,3-TCB, 1,2,3,4-TeCB, and PeCB, at 2.0–2.5 mmol/kg fish. The toxicity of chlorobenzenes to flounder (*Platichthys flesus*) and sole (*Solea solea*) increased with increasing chlorination and was similar in each species (Furay & Smith, 1995). LC₅₀ values for sheepshead minnow (*Cyprinodon variegatus*) exposed to nominal concentrations of chlorinated benzenes were similar for 48-h, 72-h, and 96-h exposure times. This was especially true for MCB and dichlorobenzenes, suggesting volatilization of the test substance (Heitmuller et al., 1981).

7.2 Terrestrial environment

The toxicity of chlorobenzenes to a terrestrial bacterium modified with the *lux* gene (*Pseudomonas fluorescens*) increased with increasing chlorination. Toxicity also increased with increasing molecular symmetry. EC₅₀ values for inhibited bioluminescence ranged from 0.57 to 118.5 mg/litre for 10 chlorobenzene congeners (Boyd et al., 1998).

EC₅₀ values for growth of lettuce (*Lactuca sativa*) exposed to chlorobenzenes ranged from 2 to 1000 mg/kg soil in a 7-day test and from 1 to >1000 mg/kg soil in a 14-day test. Toxicity increased with increasing chlorination, up to tetrachlorobenzenes. PeCB was less toxic than all congeners, with the exception of MCB (Hulzebos et al., 1993). LC₅₀ values expressed as the test solution ranged from 0.028 to 9.3 mg/litre.

The toxicity of chlorobenzenes to earthworms is presented in Table 5. LC₅₀ values for the earthworms *Eisenia andrei* and *Lumbricus rubellus* range from 75 mg/kg soil for tetrachlorobenzene (isomer not specified) to 1107 mg/kg soil for MCB; however, when the results are expressed as pore water concentrations, the values range from 0.22 µmol/litre for PeCB to 4281 µmol/litre for MCB. Two other species tested appear to be more sensitive, with LC₅₀ values of 0.0016–0.0018 mg/kg soil for 1,2,4-TCB; however, the soil type and pore water concentrations were not given, making a direct comparison with the other earthworm tests difficult.

Table 5: Toxicity of chlorobenzenes to earthworms.

Species	End-point	Medium ^a	Chlorobenzene ^b	Concentration in soil (mg/kg)	Concentration in pore water (µmol/litre)	Reference
<i>Eisenia andrei</i>	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	MCB	240	1453	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	MCB	446	797	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	DCB*	128	121	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	DCB*	229	347	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	TeCB*	75	1.6	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	TeCB*	233	1.2	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	PeCB	134	0.47	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	PeCB	238	0.25	Van Gestel et al. (1991)
<i>Lumbricus rubellus</i>	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	MCB	547	4281	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	MCB	1107	2243	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	DCB*	184	178	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	DCB*	615	1556	Van Gestel et al. (1991)

	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	TeCB*	112	2.3	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	TeCB*	201	1.1	Van Gestel et al. (1991)
	LC ₅₀	Sandy soil (3.7% OM, pH 4.8)	PeCB	115	0.43	Van Gestel et al. (1991)
	LC ₅₀	Artificial soil (8.1% OM, pH 5.9)	PeCB	201	0.22	Van Gestel et al. (1991)
<i>Eudrilus eugeniae</i>	LC ₅₀	Soil	1,2,4-TCB	0.0016		Callahan et al. (1994)
<i>Allolobaphora tuberculata</i>	LC ₅₀	Soil	1,2,4-TCB	0.0018		Callahan et al. (1994)

^a OM = organic matter.

^b Asterisk (*) indicates isomer not specified.

There is a paucity of data on the toxicity of chlorobenzenes to other terrestrial organisms.

8. EFFECTS EVALUATION

Chlorobenzenes are released to the environment during manufacture or use as an intermediate in the production of other chemicals, as a solvent, or as a degreasing agent. Some chlorobenzenes will be released directly to the environment as a result of their use as carriers for pesticides or as room deodorizers.

Chlorobenzenes released to the environment are likely to be volatilized to the atmosphere, although sorption to soils and sediment may also occur. Chlorobenzenes in the atmosphere will be degraded via photochemical oxidation reactions with hydroxyl radicals. Chlorobenzenes in the aquatic and terrestrial environment will be biodegraded, although they may persist under anaerobic conditions.

Very wide ranges of acute toxicity values have been reported for all types of aquatic organisms exposed to the various chlorobenzene congeners. These are summarized in Figure 1. Care must be taken when interpreting these data, as chlorobenzenes may volatilize from the test system (with the lower chlorinated compounds generally more volatile), thus reducing actual exposure. EC₅₀ values for microorganisms and invertebrates range from 8 to 235 000 µg/litre and from 10 to >530 000 µg/litre, respectively. LC₅₀ values for fish range from 135 µg/litre upwards. The toxicity of chlorobenzenes to aquatic organisms increases with increasing chlorination, increasing by over an order of magnitude over the chlorination range (Figure 2). This is partly due to increased uptake and bioaccumulation of higher chlorinated compounds.

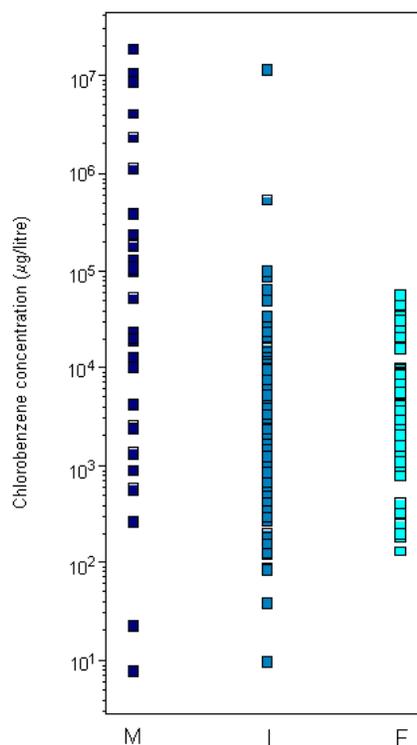


Fig. 1: Acute toxicity of chlorobenzenes (all congeners) to microorganisms (M), invertebrates (I), and fish (F) (data are from Table 3)

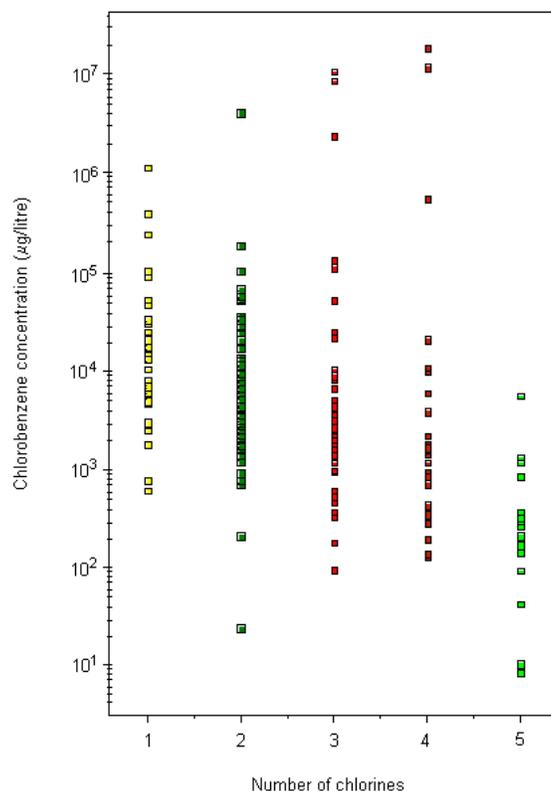


Fig. 2: Acute toxicity of chlorobenzenes to all organisms related to the chlorination of the congeners

Chronic toxicity tests are available for freshwater organisms (algae, invertebrates, and fish). Lowest NOECs, together with the organism and end-point, are presented in Table 6. Long-term studies of marine organisms reporting NOEC values are available for an invertebrate and a fish species.

Table 6: Calculation of risk factors for aquatic organisms.

	MCB	DCBs	TCBs	TeCBs	PeCB
Freshwater					
Lowest freshwater chronic NOEC (µg/litre)	1004 (daphnid reproduction)	550 (daphnid reproduction) ^a	182 (daphnid reproduction)	55 (daphnid reproduction)	32 (daphnid reproduction)
Freshwater PNEC ^b (µg/litre)	20	10	4	1	0.6
Highest measured concentration in fresh water (µg/litre) ^c	>10	>10	8.1	200	0.0006
Risk factor ^d (concentration in fresh water/PNEC)	0.5	1	2	200	0.001
Marine					
Lowest marine chronic NOEC (µg/litre)	125 (crab growth)	31 (crab growth)	25 (crab growth)	17 (crab growth)	5 (crab growth)
Marine/estuarine PNEC ^b (µg/litre)	3	0.6	0.5	0.3	0.1

Highest measured concentration in seawater ($\mu\text{g/litre}$) ^c	0.5	0.8	5.5	0.02	0.04
Risk factor ^d (concentration in seawater/PNEC)	0.2	1	10	0.07	0.4

^a Fourteen-day EC_{50} value used in the absence of a suitable NOEC.

^b PNEC is calculated by dividing the lowest chronic NOEC by an uncertainty factor of 50 and rounding the calculated value to one significant figure.

^c Concentration values were the highest mean values reported in individual studies. Data from Yamamoto et al. (1997) were not included in this risk assessment, as it was unclear whether the water samples analysed were fresh, estuarine, or marine. Where data were reported as $>10 \mu\text{g/litre}$, a value of $10 \mu\text{g/litre}$ was used to calculate the risk factor.

^d Risk factors are rounded to one significant figure.

For compounds studied as extensively as the chlorobenzenes in fresh water, an uncertainty factor of 10 could normally be applied to generate a predicted no-effect concentration (PNEC) for aquatic organisms. However, the lack of chronic NOEC data for algae requires the conservative application of an uncertainty factor of 50. This has been done for both the freshwater and marine data, and results are presented in Table 6.

The concentrations of chlorobenzenes measured in surface waters are presented in Figure 3. This figure includes data points from some additional studies that are not discussed in the text. However, the concentrations in these existing studies fall within the range of concentrations that are discussed in the text. In the 1970s, MCB and dichlorobenzene concentrations in fresh waters ranged up to $10 \mu\text{g/litre}$, and trichlorobenzene concentrations ranged up to $1 \mu\text{g/litre}$ (IPCS, 1991a). Chlorobenzene concentrations measured since the late 1980s show a maximum of $0.6 \mu\text{g/litre}$ for MCB, a maximum of $130 \mu\text{g/litre}$ for dichlorobenzenes, and a maximum of $10 \mu\text{g/litre}$ for trichlorobenzenes (Figure 3). Tetrachlorobenzene concentrations in fresh water show a maximum of $200 \mu\text{g/litre}$, and PeCB, $0.0006 \mu\text{g/litre}$; these are older measurements (Figure 3).

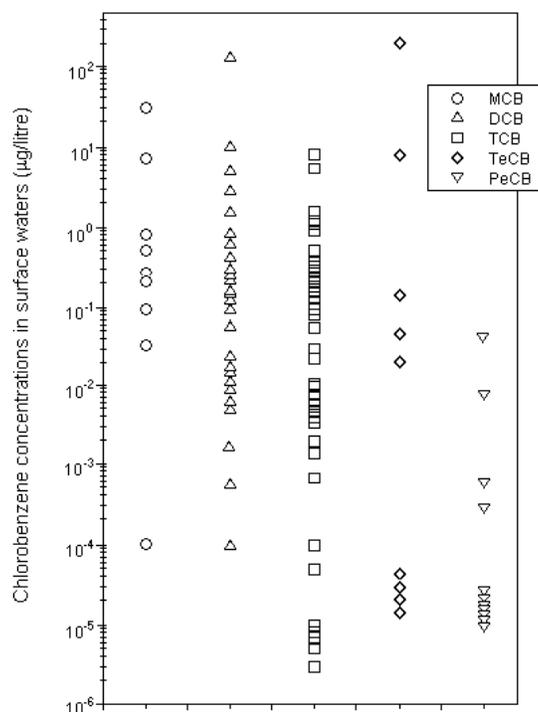


Fig. 3: Concentrations of chlorobenzene congeners in surface waters

Lowest chronic NOECs, PNECs, highest measured water concentrations, and risk factors are summarized in Table 6. Risk factors at, below, or substantially below 1 are generated for all chlorinated benzene congeners with the exception of trichlorobenzenes and tetrachlorobenzenes. Factors of 2 and 10 for trichlorobenzenes in fresh water and seawater, respectively, and a factor of 200 for tetrachlorobenzene in fresh water indicate some risk, particularly for trichlorobenzenes in seawater and tetrachlorobenzenes in fresh water, with the use of the precautionary uncertainty factor of 50.

The data set on freshwater concentrations of chlorobenzenes is limited, and the high risk factors for trichlorobenzenes and tetrachlorobenzenes come from monitoring studies carried out during the early to mid-1980s. The trichlorobenzene concentrations were reported in a single study of Spanish rivers in industrial areas conducted in the mid-1980s. The highest reported concentrations come from one small river with a low flow rate. The highest tetrachlorobenzene concentrations were reported from areas near dump sites. No follow-up monitoring is available for either of these sites. The only conclusion that can be drawn is that under these exceptional circumstances, concentrations of trichlorobenzenes can exceed those likely to produce long-term, but not acute, toxic effects; it can be presumed that these concentrations would arise from point sources. More generally, measured levels of trichlorobenzenes are substantially lower than this, and risk factors would be below 1.

There are more extensive measurements of chlorobenzenes in estuaries and coastal waters. The risk factors in estuarine water exceeding 1 relate to point sources from industrial plants pre-1989. Follow-up monitoring is available for these plants following control of release and/or replacement of chlorobenzenes with alternatives. These more recent data indicate trichlorobenzene concentrations between 1 and 2 orders of magnitude lower. As with fresh water, uncontrolled point source release of trichlorobenzenes will lead to high local risk to organisms.

Terrestrial data, both toxicity studies and measured levels in soil, are inadequate to perform a risk assessment.

9. PREVIOUS EVALUATIONS BY INTERNATIONAL BODIES

Chlorinated benzenes were reviewed by the World Health Organization in 1991 (IPCS, 1991a).

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APPENDIX 1 — SOURCE DOCUMENT

IPCS (1991a) *Chlorobenzenes other than hexachlorobenzene*. Geneva, World Health Organization, International Programme on Chemical Safety (Environmental Health Criteria 128)

A WHO Task Group on Environmental Health Criteria for Chlorobenzenes Other than Hexachlorobenzene met at the Institut d'Hygiène et d'Epidémiologie, Brussels, Belgium, from 25 to 29 June 1990. The Task Group reviewed and revised the draft criteria document and made an evaluation of

the risks for human health and the environment from exposure to chlorobenzenes other than hexachlorobenzene. The drafts of this document were prepared by Ms M.E. Meek and Ms M.J. Giddings, Health Protection Branch, Health and Welfare Canada, Ottawa, Canada. Dr G.C. Becking, IPCS Interregional Research Unit, WHO, Research Triangle Park, NC, USA, was responsible for the overall scientific content of the document, and Mrs M.O. Head, Oxford, England, for the editing. Extensive comments were received from Dr U. Schlottmann, Federal Ministry of the Environment, Germany (chemistry and environmental effects), and Dr R. Fielder, Department of Health, United Kingdom (effects on experimental animals), during the initial review of the document. Dr S. Dobson, Co-Chairman of the Task Group, and Dr P.E.T. Douben made significant contributions and revisions of the draft document during the meeting, particularly the sections dealing with environmental effects.

This CICAD was prepared with reference to the above source document. Additional information from other national assessments was also included:

ATSDR (1990) *Toxicological profile for chlorobenzene*. Atlanta, GA, US Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry.

ATSDR (1998) *Toxicological profile for 1,4-dichlorobenzene (update)*. Atlanta, GA, US Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry.

For more information on these ATSDR assessments, contact:

Agency for Toxic Substances and Disease Registry
Division of Toxicology
1600 Clifton Road NE, Mailstop E-29
Atlanta, GA 30333
USA
Telephone: 1-888-422-8737
Fax: 404-498-0057

APPENDIX 2 — CICAD PEER REVIEW

The draft CICAD on chlorobenzenes other than hexachlorobenzene (environmental aspects) was sent for review to institutions and organizations identified by IPCS after contact with IPCS national Contact Points and Participating Institutions, as well as to identified experts. Comments were received from:

- M. Baril, Institut de Recherche en Santé et en Sécurité du Travail, Montreal, Canada
- R. Benson, Drinking Water Program, US Environmental Protection Agency, Denver, CO, USA
- P. Copestake, Toxicology Advice & Consulting Ltd, Surrey, United Kingdom
- I. Desi, University of Szeged, Szeged, Hungary
- E. Frantik, National Institute of Public Health, Prague, Czech Republic
- A. Juhasz, University of South Australia, Mawson Lakes, Australia
- U. Kierdorf, Justus-Liebig-University of Giessen, Giessen, Germany
- S. Schmidt, Fraunhofer Institute for Toxicology and Experimental Medicine, Hanover, Germany

APPENDIX 3 — CICAD FINAL REVIEW BOARD

Varna, Bulgaria
8–11 September 2003

Members

- Dr I. Benchev, Sofia, Bulgaria
- Dr R. Chhabra, National Institute of Environmental Health Sciences, Research Triangle Park, NC, USA
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- Dr L. Fishbein, Fairfax, VA, USA
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- Dr R.F. Hertel, Federal Institute for Risk Assessment, Berlin, Germany
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APPENDIX 4 — ABBREVIATIONS AND ACRONYMS

ATSDR	Agency for Toxic Substances and Disease Registry (USA)
BCF	bioconcentration factor
CICAD	Concise International Chemical Assessment Document
DCB	dichlorobenzene
EC ₅₀	median effective concentration
ECD	electron capture detector
EHC	Environmental Health Criteria
FID	flame ionization detector
GC	gas chromatography
HDTMA	hexadecyltrimethylamminium
ICSC	International Chemical Safety Card
ILO	International Labour Organization
IPCS	International Programme on Chemical Safety
K _{oa}	octanol/air partition coefficient
	soil sorption coefficient; suspended sediment/water partition coefficient

K_{oc}	
K_{ow}	octanol/water partition coefficient
LC ₅₀	median lethal concentration
LOEC	lowest-observed-effect concentration
MCB	monochlorobenzene
MS	mass spectrometry
NOEC	no-observed-effect concentration
OECD	Organisation for Economic Co-operation and Development
PeCB	pentachlorobenzene
PIM	Poison Information Monograph
PNEC	predicted no-effect concentration
SD	standard deviation
TCB	trichlorobenzene
TeCB	tetrachlorobenzene
TRI	Toxics Release Inventory (USA)
UNEP	United Nations Environment Programme
USA	United States of America
V_{max}	maximum rate of reaction
WHO	World Health Organization

INTERNATIONAL CHEMICAL SAFETY CARDS

[CHLOROGENZENE ICSC:0642](#)

[1,2-DICHLOROGENZENE ICSC:1066](#)

[1,3-DICHLOROGENZENE ICSC:1095](#)

[1,4-DICHLOROGENZENE ICSC:0037](#)

[1,2,3-TRICHLOROGENZENE ICSC:1222](#)

[1,2,4-TRICHLOROGENZENE ICSC:1049](#)

[1,3,5-TRICHLOROGENZENE ICSC:0344](#)

[1,2,4,5-TETRACHLOROGENZENE ICSC:0676](#)

[PENTACHLOROGENZENE ICSC:0531](#)

RÉSUMÉ D'ORIENTATION

Le présent CICAD consacré aux chlorobenzènes autres que l'hexachlorobenzène (aspects environnementaux) est une mise à jour du No 128 de la série Critères d'hygiène de l'environnement (CHE) intitulé *Chlorobenzènes autres que l'hexachlorobenzène* (IPCS, 1991a). Des renseignements sur le devenir et la concentration des chlorobenzènes ont également été tirés de rapports sur le chlorobenzène (ATSDR, 1990) et le 1,4-dichlorobenzène (ATSDR, 1998) publiés par l'Agency for Toxic Substances and Disease Registry. On a poursuivi le dépouillement de la littérature jusqu'en décembre 2002 à la recherche de références complémentaires qui auraient été publiés postérieurement à ces mises au point. L'appendice 1 donne des informations sur la nature de l'examen par des pairs du document initial. Des renseignements sur l'examen par des pairs du présent CICAD sont donnés à l'appendice 2. Ce CICAD a été adopté en tant qu'évaluation internationale lors de la réunion du Comité d'évaluation finale qui s'est tenue à Varna (Bulgarie) du 8 au 11 septembre 2003. La liste des participants à cette réunion figure à l'appendice 3. Les fiches internationales sur la sécurité chimique de divers chlorobenzènes (ICSC 0037, 0344, 0531, 0642, 0676, 1049, 1066, 1095, 1222), établies par le Programme international sur la sécurité chimique (IPCS, 2000, 2003a-h), sont également reproduites dans le présent document. Ce CICAD porte principalement sur les aspects environnementaux de ces composés car depuis la publication du CHE (IPCS, 1991a), l'évaluation des risques pour la santé humaine n'a pas sensiblement changé.

Les dérivés chlorés du benzène ou chlorobenzènes constituent un groupe de composés aromatiques cycliques dans lesquels un atome de chlore est substitué à un ou plusieurs atomes d'hydrogène du noyau benzénique. On les utilise principalement comme intermédiaires dans la synthèse de pesticides ou d'autres produits chimiques. Le 1,4-dichlorobenzène (1,4-DCB) est utilisé comme désodorisant d'ambiance et comme antimites. Les dérivés plus substitués (comme les trichlorobenzènes, le 1,2,3,4-tétrachlorobenzène [1,2,3,4-TeCB] et le pentachlorobenzène [PeCB]) ont été utilisés comme fluides diélectriques.

On n'a pas découvert de sources naturelles de chlorobenzènes. Des chlorobenzènes sont rejetés dans l'environnement lors de leur préparation ou lorsqu'on les utilise comme intermédiaires pour la synthèse d'autres produits. Ils passent également dans l'environnement lorsque des déchets qui en contiennent sont éliminés, par exemple par incinération ou décharge sur des sites mal sécurisés. Le monochlorobenzène (MCB) est rejeté directement dans le milieu ambiant lors de son utilisation comme charge dans certains pesticides. Le passage dans l'environnement des chlorobenzènes utilisés comme désodorisants, fumigants, dégraissants, insecticides, herbicides et défoliants est également la conséquence directe de leur utilisation.

Compte tenu de leurs propriétés physicochimiques, les chlorobenzènes rejetés dans l'environnement se vaporisent vraisemblablement dans l'atmosphère. Ils s'en éliminent ensuite principalement après avoir réagi sur les radicaux hydroxyles pour donner du nitrochlorobenzène, du chlorophénol et des dérivés aliphatiques dicarboxylés, à leur tour éliminés par photolyse ou réaction avec ces mêmes radicaux hydroxyles. Les chlorobenzènes qui passent dans l'environnement aquatique se redistribuent préférentiellement dans l'air et dans les sédiments (notamment les sédiments riches en matières organiques). En solution aqueuse, les chlorobenzènes pourraient théoriquement subir une déchloration réductrice par voie photochimique, mais cette possibilité a été étudiée dans des conditions artificielles qui n'étaient pas représentatives des régions tempérées. Dans le sol, c'est le phénomène de sorption qui constitue le facteur le plus important dont dépendent le devenir et le comportement des chlorobenzènes. Les processus de sorption-désorption qui se déroulent dans le sol conditionnent la vitesse d'évaporation et de lessivage de ces produits, de même que leur faculté de subir une décomposition chimique ou microbienne ou encore d'être fixés par des végétaux ou d'autres organismes.

Dans les divers substrats où ils sont présents - sol, sédiments ou boues d'égout -, les chlorobenzènes sont susceptibles de subir une biodégradation microbienne. Le principal mécanisme de la décomposition anaérobie comporte une déchloration oxydative conduisant à la formation de composés aromatiques hydroxylés (essentiellement des catéchols) suivie de l'ouverture du cycle et d'une minéralisation en dioxyde de carbone et eau. Les chlorobenzènes les moins substitués sont plus facilement dégradés que les autres.

La bioaccumulation de chlorobenzènes par les organismes aquatiques dépend de l'hydro- et de la liposolubilité relative de ces produits (qui correspond au coefficient de partage octanol/eau) ainsi que du degré de substitution. Ces composés sont d'autant mieux captés dans l'eau qu'ils sont plus substitués et que la température est plus élevée.

On a décelé la présence de chlorobenzènes (MCB, dichlorobenzènes et trichlorobenzènes) dans l'air ambiant à une concentration moyenne de 0,1 µg/m³, avec des maxima pouvant atteindre 100 µg/m³ dans des sites dangereux. Dans les eaux de surface, la concentration des chlorobenzènes oscille généralement entre des valeurs de l'ordre du nanogramme ou du microgramme par litre, avec des maxima allant jusqu'à 0,2 mg/litre à proximité des sites industriels. La teneur en chlorobenzènes des eaux usées industrielles est sans doute plus élevée et varie selon les procédés industriels mis en oeuvre. Dans les sols non contaminés, la concentration des différents dichlorobenzènes est généralement inférieure à 0,4 mg/kg et celle des autres chlorobenzènes inférieure à 0,1 mg/kg. Dans les sédiments, les teneurs se situent habituellement dans une gamme de valeurs allant du ng/kg au µg/kg, encore que des valeurs de l'ordre du mg/kg aient été observées dans des échantillons prélevés sur des sites industriels.

D'une façon générale, la toxicité pour les organismes aquatiques augmente avec le degré de chloration du noyau benzénique. Les valeurs de la CE₅₀ à 72 h pour les algues vertes vont de 5280 µg/litre dans le cas du 1,3-DCB à 200 000 µg/litre dans le cas du MCB; de même, la CE₅₀ à 48 h pour les diatomées va de 8 à 235 000 µg/litre. En ce qui concerne les invertébrés d'eau douce, les valeurs de la CE₅₀ à 48 h sont comprises entre 10 µg/litre dans le cas du PeCB et >530 000 µg/litre dans le cas du 1,2,4,5-TeCB. Pour les poissons, les valeurs de la CL₅₀ à 96 h vont de 135 µg/litre dans le cas du PeCB à 21 000 µg/litre dans le cas du 1,2,4-trichlorobenzène (1,2,4-TCB). Les concentrations chroniques sans effet observable (NOEC) sur les invertébrés d'eau douce oscillent entre 32 µg/litre dans le cas du PeCB et 19 000 µg/litre dans le cas du MCB; pour les poissons, les valeurs de la NOEC vont de 18 µg/litre avec le PeCB à 8 500 µg/litre avec le MCB.

On ne possède guère de données concernant les effets des chlorobenzènes sur les organismes terrestres. Pour les plantes en cultures hydroponiques ou en cultures sur sol, on a trouvé des valeurs de la CL₅₀ allant respectivement de 0,028 à 9,3 mg/litre et de 1 à plus de 1000 mg/kg de sol. Pour les lombrics *Eisenia andrei* et *Lumbricus rubellus*, on a mesuré des valeurs de la CL₅₀ qui sont comprises entre 0,22 µmol/litre (eau des pores) dans le cas du PeCB et 4281 µmol/litre dans le cas du MCB.

Il y a peu de chances que les chlorobenzènes aient des effets nocifs sur les organismes aquatiques. Les facteurs de risque obtenus en comparant les valeurs de la toxicité chronique aux concentrations mesurées dans l'environnement sont généralement inférieurs à 1, sauf pour quelques composés pour lesquels ils sont plus élevés, avec la valeur maximum de 200. Les facteurs de risque les plus élevés ont été obtenus en utilisant d'anciennes données portant sur des sources ponctuelles et ne sont donc pas représentatifs de l'environnement dans son ensemble, notamment si l'on tient compte de l'évaporation. On ne disposait pas de données suffisantes pour évaluer le risque auquel sont exposées les espèces terrestres.

RESUMEN DE ORIENTACIÓN

Este CICAD sobre los clorobencenos distintos del hexaclorobenceno (aspectos ecológicos) es una actualización del N° 128 de los Criterios de Salud Ambiental (EHC), *Chlorobenzenes other than hexachlorobenzene* (IPCS, 1991a). También se obtuvo información sobre el destino y las concentraciones de los clorobencenos a partir de los informes de la Agencia para el Registro de Sustancias Tóxicas y Enfermedades sobre el clorobenceno (ATSDR, 1990) y el 1,4-diclorobenceno (ATSDR, 1998). Se realizó asimismo una búsqueda bibliográfica hasta diciembre de 2002 para identificar cualquier información que se hubiera publicado después de la terminación de dichos informes. La información sobre el examen colegiado del documento original se presenta en el apéndice 1. La información sobre el examen colegiado de este CICAD aparece en el apéndice 2. Este CICAD se aprobó como evaluación internacional en una reunión de la Junta de Evaluación Final celebrada en Varna (Bulgaria) del 8 al 11 de septiembre de 2003. La lista de participantes en esta reunión figura en el apéndice 3. También se reproducen en este documento las Fichas internacionales de seguridad química para varios clorobencenos distintos (ICSC 0037, 0344, 0531, 0642, 0676, 1049, 1066, 1095, 1222), preparadas por el Programa Internacional de Seguridad de las Sustancias Químicas (IPCS, 2000, 2003a-h). Este CICAD se concentra en los aspectos ecológicos porque no se han registrado cambios significativos para la salud humana desde la publicación de los Criterios de Salud Ambiental (IPCS, 1991a).

Los bencenos clorados son un grupo de compuestos aromáticos cíclicos en los cuales se han sustituido uno o más átomos de hidrógeno del anillo de benceno por un átomo de cloro. Los clorobencenos se utilizan principalmente como intermediarios en la síntesis de plaguicidas y otros productos químicos. El 1,4-diclorobenceno (1,4-DCB) se utiliza en desodorantes ambientales y como repelente de la polilla. Los bencenos más clorados (trichlorobencenos, 1,2,3,4-tetraclorobenceno [1,2,3,4-TeCB] y pentaclorobenceno [PeCB]) se han utilizado como componentes de fluidos dieléctricos.

No se han identificado fuentes naturales de clorobencenos en el medio ambiente. Los clorobencenos se liberan en el medio ambiente durante su fabricación o su utilización como intermediarios en la producción de otras sustancias químicas. También hay emisiones durante la eliminación de productos de

clorobenceno, por ejemplo en los incineradores y en los vertederos de desechos peligrosos. El monoclorobenceno (MCB) se libera directamente en el medio ambiente debido a su utilización como excipiente de plaguicidas. Los clorobencenos que se utilizan como desodorantes, fumigantes, desengrasantes, insecticidas, herbicidas y defoliantes también se liberan en el medio ambiente como resultado directo de su aplicación.

Sus propiedades fisicoquímicas parecen indicar que es probable que los clorobencenos liberados en el medio ambiente se volatilicen en la atmósfera. La eliminación de la atmósfera se produce fundamentalmente por medio de su reacción con radicales hidroxilo para formar nitroclorobenceno, clorofenol y dicarbonilos alifáticos, que posteriormente se degradan por fotólisis o por reacción con radicales hidroxilo. Los clorobencenos liberados en el medio acuático se redistribuyen preferentemente entre el aire y los sedimentos (sobre todo los que tienen abundante materia orgánica). En teoría, los clorobencenos en soluciones acuosas podrían sufrir una dechloración fotoquímica reductora, aunque sólo se han realizado estudios en condiciones artificiales que no eran representativas de las regiones templadas. El factor más importante que afecta al comportamiento y el destino de los clorobencenos en el suelo es la sorción. Los procesos de adsorción y desorción en el suelo afectan a la tasa de volatilización y lixiviación y a la disponibilidad de sustancias químicas para la degradación microbiana y química o para la absorción por las plantas u otros organismos.

Los microorganismos pueden degradar los clorobencenos en distintos sustratos, por ejemplo el suelo, los sedimentos y los lodos cloacales. El principal mecanismo de degradación aerobia es la dechloración oxidativa, que da lugar a la formación de compuestos aromáticos hidroxilados (sobre todo catecoles), cuyo anillo se rompe y se produce la mineralización posterior hasta dióxido de carbono y agua. Los bencenos menos clorados se degradan más fácilmente que los más clorados.

La bioacumulación de clorobencenos por los organismos acuáticos depende de su solubilidad relativa en el agua y los lípidos (reflejando de esta manera los coeficientes de reparto octanol/agua) y del número de sustituciones de cloro. La absorción a partir del agua aumenta con el grado de cloración y la temperatura.

Se ha notificado la presencia de clorobencenos (MCB, diclorobencenos y triclorobencenos) en el aire exterior, con concentraciones medias del orden de $0,1 \mu\text{g}/\text{m}^3$ y máximas (en vertederos de desechos peligrosos) de hasta $100 \mu\text{g}/\text{m}^3$. Las concentraciones de clorobencenos en las aguas superficiales suelen estar en la escala de ng/l a $\mu\text{g}/\text{l}$, con concentraciones máximas de hasta $0,2 \text{ mg}/\text{l}$ en zonas próximas a fuentes industriales. Las concentraciones de clorobencenos en las aguas residuales industriales pueden ser más elevadas y variar en función del tipo de proceso utilizado. Sus concentraciones en suelos no contaminados son en general inferiores a $0,4 \text{ mg}/\text{kg}$ de diclorobencenos y inferiores a $0,1 \text{ mg}/\text{kg}$ para otros clorobencenos. Las concentraciones de clorobencenos en los alimentos suelen estar en la escala de ng/kg a $\mu\text{g}/\text{kg}$, aunque se ha informado de concentraciones del orden de mg/kg en muestras procedentes de zonas industriales.

En general, la toxicidad en el medio acuático aumenta con el grado de cloración del anillo de benceno. La CE_{50} a las 72 h para las algas verdes oscila entre $5280 \mu\text{g}/\text{l}$ para el 1,3-DCB y $200\,000 \mu\text{g}/\text{l}$ para el MCB; de igual forma, la CE_{50} a las 48 h para las diatomeas varía entre 8 y $235\,000 \mu\text{g}/\text{l}$. En los invertebrados de agua dulce, la CE_{50} a las 48 h oscila entre $10 \mu\text{g}/\text{l}$ para el PeCB y $> 530\,000 \mu\text{g}/\text{l}$ para el 1,2,4,5-TeCB. La CL_{50} a las 96 h para los peces varía entre $135 \mu\text{g}/\text{l}$ para el PeCB y $21\,000 \mu\text{g}/\text{l}$ para el 1,2,4-triclorobenceno (1,2,4-TCB). Las concentraciones sin efectos crónicos observados (NOEC) para los invertebrados de agua dulce oscilan entre $32 \mu\text{g}/\text{l}$ para el PeCB y $19\,000 \mu\text{g}/\text{l}$ para el MCB; en los peces, las NOEC varían entre $18 \mu\text{g}/\text{l}$ para el PeCB y $8500 \mu\text{g}/\text{l}$ para el MCB.

Hay pocos datos disponibles sobre los efectos de los clorobencenos en los sistemas terrestres. Los valores de las CL_{50} para las plantas de cultivos hidropónicos y cultivadas en el suelo oscilan entre $0,028$ y $9,3 \text{ mg}/\text{l}$ y entre 1 y $> 1000 \text{ mg}/\text{kg}$ de suelo, respectivamente. Los valores de las CL_{50} para las lombrices *Eisenia andrei* y *Lumbricus rubellus* varían entre $0,22 \mu\text{moles}/\text{l}$ (agua intersticial) para el PeCB y $4281 \mu\text{moles}/\text{l}$ para el MCB.

El riesgo de que los bencenos clorados provoquen daños en los organismos acuáticos es bajo. Los factores de riesgo comparando los valores de la toxicidad crónica con las concentraciones medidas en el medio ambiente son en general inferiores a 1, salvo algunos compuestos que tenían factores de riesgo más altos, con un valor máximo de 200. Los factores de riesgo más elevados se obtuvieron utilizando antiguos datos de fuentes puntuales, por lo que no son representativos del medio en su conjunto, en particular cuando se tiene en cuenta la probabilidad de evaporación. No se dispuso de datos suficientes para realizar una evaluación del riesgo en las especies terrestres.

ENDNOTES:

1. International Programme on Chemical Safety (1994) *Assessing human health risks of chemicals: derivation of guidance values for health-based exposure limits*. Geneva, World Health Organization (Environmental Health Criteria 170) (also available at <http://www.who.int/pcs/>).

See Also:

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Pentachlorobenzene

Type: ○

CAS No. 608935

Physical Properties

	Value	Reference
Molecular Weight (g/mol)	250.33894	TR
Solubility @ 20-25 degC (mg/L)	0.65	TR
Vapor pressure @ 20-25 degC (mmHG)	0.001672	TR
Henry's Law constant @ 20 degC	0.0315932	TR
Sorption coefficient (log L/kg) K _{oc}	4.5	TR
Octanol-water partition coefficient (log L/kg)	5.2155	TR
Diffusion coefficient in air (cm ² /s)	0.067	TR
Diffusion coefficient in water (cm ² /s)	6.3e-006	TR

Miscellaneous Parameters

	Value	Reference
Relative bioavailability factor (-)	1	T
Analytical Detection Limits:		
Water (mg/L)	-	-
Soil (mg/kg)	-	-
First-order decay half lives (days):		
Saturated zone	-	-
Unsaturated zone	-	-
Soil-to-plant biotransfer factor (0):		
Above ground veg.	-	-

LAST UPDATE:

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