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CANADIAN ENVIRONMENTAL PROTECTION ACT

PRIORITY SUBSTANCES LIST

SUPPORTING DOCUMENT

ENVIRONMENTAL SECTIONS

1,4-DICHLOROBENZENE

Government of Canada
Environment Canada
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1.0 IDENTITY OF SUBSTANCE

1.1 Name of Substance

IUPAC Name: 1,4-Dichlorobenzene

Synonyms and Trade Names:
    p-dichlorobenzene
    p-dichlorobenzenol
    paradichlorobenzene
    para-dichlorobenzene
    para-dichlorobenzenol
    1,4-dichlorobenzene
    PDB
    PDCB
    Di-chloricide
    Paracide
    Paradi
    Paradow
    Paramoth
    Santochlor
    Parazene
    Paranuggets
    paraCrystals
    p-chlorophenyl chloride
    Evola
    Persia-Perazol

CAS Number: 106-46-7

PSL Name: 1,4-Dichlorobenzene

1.2 Characteristics of Substance

1,4-Dichlorobenzene is a neutral, colourless, flammable solid at 25°C, with an aromatic odour (U.S. EPA, 1986). 1,4-Dichlorobenzene is a cyclic aromatic compound with two hydrogen atoms on the benzene ring substituted with chlorine. The molecular weight of 1,4-dichlorobenzene is 147.01 g/mol and the empirical formula is $C_6H_4Cl_2$. The structure of 1,4-dichlorobenzene is presented in Figure 1-1.

1,4-Dichlorobenzene contains less than 0.1% monochlorobenzene and trichlorobenzenes, and less than 0.5% each of 1,2- and 1,3-dichlorobenzene (Kao and Poffenberger, 1979).

1.3 Analytical Methodology

Analytical methods used to quantify 1,4-dichlorobenzene in environmental media include gas chromatography with flame ionization or electron capture detection and gas

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chromatography/mass spectrometry (Oliver and Nicol, 1984, Oliver and Bothen, 1982).

2.0 PHYSICAL AND CHEMICAL PROPERTIES

The physical and chemical characteristics affecting the environmental fate and toxicity of 1,4-dichlorobenzene are presented in Table 2-1. It has moderate vapour pressure (90 Pa @ 25°C), a moderate octanol/water partition coefficient (log K_{ow} = 3.4) and a low water solubility (79 mg/L @ 25°C).

3.0 SOURCES AND RELEASES TO THE ENVIRONMENT

3.1 Natural Sources

No natural sources of 1,4-dichlorobenzene have been identified.

3.2 Anthropogenic Sources

3.2.1 Quantities in Use

Production Volume and Trends

1,4-Dichlorobenzene is produced in Canada. Annual production of total dichlorobenzenes (1,2- and 1,4-dichlorobenzene) between 1987 and 1990, averaged between 5 kilo tonnes and 6 kilo tonnes. The production value is actually lower as it includes imported crude material which the company processes further once in Canada. Currently, one Canadian producer manufactures both 1,2-dichlorobenzene and 1,4-dichlorobenzene. From 1989 to 1991, the number of companies which were listed as chemical suppliers in Canada decreased from seven to three (Process Industries Canada, 1989 to 1991). The Canadian demand for 1,4-dichlorobenzene remained steady from 1985 to 1990 (Table 3-1), averaging approximately 3 500 tonnes per year, and this is not expected to change until 1995 (Camford Information Services, 1991).

In the United States, seven companies produced an estimated 27 500 tonnes of 1,4-dichlorobenzene in 1978 (U.S. EPA, 1982). In 1990, three companies reported a production capacity of 62 600 tonnes of 1,4-dichlorobenzene (Chemical Marketing Reporter, 1990). The demand for 1,4-dichlorobenzene in the U.S. in 1990 was 36 300 tonnes.

3.2.2 Imports and Exports

Volume of Importation

The total amount of 1,4-dichlorobenzene imported into Canada was steady from 1986 to 1990, averaging about 2 785 tonnes per year

(Table 3-2). In 1989, 1,4-dichlorobenzene was imported from the United States, Brazil and West Germany (Environment Canada, 1990, unpubl.).

Volume of Exportation

In terms of trade, 1,4-dichlorobenzene accounts for approximately 80% of all dichlorobenzene imports and the bulk of Canada's exports. The United States, Asia, South America and Central America are key export markets.

3.2.3   Manufacturing Processes

1,4-Dichlorobenzene is produced by chlorinating benzene in the presence of a Friedel-Crafts catalyst (Kao and Poffenberger, 1979). The usual catalyst is ferric chloride, either as such or generated in situ by exposing a large surface of iron to the liquid being chlorinated (U.S. EPA, 1986). Through the manipulation of process controls such as type of catalyst, temperature and the benzene-chlorine ratio, it is possible to maximize the production of specific chlorobenzenes. Each of the chlorinated benzenes, except hexachlorobenzene, can be chlorinated further, hence the compound is always a mixture of chlorinated benzenes. Pure compounds are obtained by distillation and crystallization.

3.2.4   Uses

In Canada, the sanitation industry is the largest domestic market for 1,4-dichlorobenzene. 1,4-Dichlorobenzene is used primarily in air fresheners, as a deodorizer in urinals, and as moth and bird repellents (Camford Information Services, 1991; Davis, 1992 pers. comm.). There are currently five plants in Canada which formulate a total of 12 commercial products which use 1,4-dichlorobenzene as the active ingredient in pesticides (Davis, 1991 pers. comm.). Data on use patterns for this substance were identified for the period 1977 to 1979 when 99.3% of the annual demand was used in the production of deodorant blocks and 0.7% in the production of moth crystals (Environment Canada, 1983 unpubl.). Camford Information Services (1991) forecasts a slight increase in the use of 1,4-dichlorobenzene from 1900 tonnes to 2500 tonnes for 1995.

In the United States, 1,4-dichlorobenzene is used extensively as a moth repellent (U.S. EPA, 1986). In 1990, 1,4-dichlorobenzene was used in space deodorants and moth control agents (40%), exported (25%), and used in the production of polyphenylene sulfide (PPS) resin (25%) and 1,2,4-trichlorobenzene (10%) (Chemical Marketing Reporter, 1990). However, the traditional uses of 1,4-dichlorobenzene, in space deodorants and moth control agents, are expected to decline in the long term whereas production of PPS resin is expected to grow (Chemical Marketing Reporter, 1990).

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3.2.5   Releases

There is limited quantitative information on anthropogenic releases of 1,4-dichlorobenzene to the Canadian environment. Because of its volatility and dispersive use patterns, it is estimated that the majority of the Canadian demand (3 500 tonnes) is released to the environment. Release of 1,4-dichlorobenzene to the environment may result from any stage in the production, transport, use and disposal of 1,4-dichlorobenzene-containing material. A summary of concentrations of 1,4-dichlorobenzene in various effluents released to the Canadian environment is summarized in Table 3-3.

Industrial and Municipal Wastewater

Releases of 1,4-dichlorobenzene in effluents from organic and inorganic chemical manufacturing plants in Ontario were reported under the Municipal/Industrial Strategy for Abatement (MISA). Under the MISA program, 1,4-dichlorobenzene was detected between October 1, 1989 and January 31, 1991, in effluents discharged from organic chemical manufacturing plants at Sarnia and Corunna, Ontario to the St. Clair River; loading values were estimated to range between 0.002 and 0.365 kg/day. Also, under the MISA program, loading values ranging between 0.001 and 0.055 kg/day, were reported for the St. Lawrence, Welland and Mary's Rivers, Ontario (OME 1992, unpubl.; OME 1992a). Loadings ranged between 0.001 to 0.005 kg/day 1,4-dichlorobenzene into the St. Mary's River, and Lake Superior, the Niagara River and Lake Ontario, from two of 28 inorganic chemical plants monitored between December 1989 and January 1991 (OME, 1992b).

Oliver and Pugsley (1986) reported that 1,4-dichlorobenzene had been released into the St. Clair River in substantial, but unquantified amounts, prior to the early 1980's. The source was related to leakage from an industrial chemical plant site and a landfill site in Sarnia, Ontario. Oliver and Charlton (1984) reported that the major source of 1,4-dichlorobenzene to Lake Ontario was from the Niagara River with an approximate loading of 23 kg/day 1,4-dichlorobenzene for the period from May to November, 1982. Chemical dumpsite leachates and direct discharges appeared to be the major sources of 1,4-dichlorobenzene to the Niagara River and Lake Ontario (Oliver, 1987a). Oliver (1987a) reported a loading value of 5 800 kg/year (~16 kg/day) 1,4-dichlorobenzene to Lake Ontario from the Niagara River for the period between September 1981 and April 1983. Based on concentrations of 1,4-dichlorobenzene in sewage plant effluents by Oliver and Nicol (1982), and total sewage treatment discharge, Oliver (1987a) was able to estimate loadings of 1.8 kg/day (660 kg/year) 1,4-dichlorobenzene to Lake Ontario. The source of 1,4-dichlorobenzene to the Niagara River was attributed to direct discharges from chemical manufacturers and over 200 chemical waste sites along the River (Oliver and Nicol, 1982). Kuntz and Wary (1983) reported

that the Niagara River was a significant source of organochlorines to Lake Ontario. Loadings of 1,4-dichlorobenzene to Lake Ontario from Niagara River suspended sediments were estimated at 0.85 kg/day (310 kg/year) for the period 1980 to 1981.

Data from sediment cores from Lake Ontario indicate that 1,4-dichlorobenzene has been accumulating in the sediments for approximately 60 years (Durham and Oliver, 1983; Oliver and Nicol, 1982). The authors demonstrated that levels of 1,4-dichlorobenzene in sediment core samples followed known production levels and use patterns for this substance in the United States. A dramatic increase in production of 1,4-dichlorobenzene occurred after the early 1940's and reached a peak during the 1960's when production of chlorobenzenes was at a maximum.

Twenty-seven pulp and paper mills which discharge directly to surface water in Ontario were also monitored under the MISA program. 1,4-Dichlorobenzene was detected in the process effluent from five mills with maximum concentrations ranging from 0.22 to 9.60 µg/L from January 1, 1990 to December 31, 1990 (OME, 1991a; 1991b). 1,4-Dichlorobenzene was also measured in five of ten Canadian textile mills sampled in a 1985/86 survey of Canadian textile mill wastewater. Concentrations of 1,4-dichlorobenzene were reported to range up to a maximum of 71.1 µg/L (Environment Canada, 1989). King and Sherbin (1986) conducted a one-day effluent survey to characterize point sources of organic chemicals in industrial, municipal and landfill effluent at the petrochemical industrial site at Sarnia, Ontario in 1985. 1,4-Dichlorobenzene was detected in landfill leachate, stormwater, and runoff from industrial, municipal and non-industrial sources prior to discharging into the St. Clair River. The Township Ditch integrates the discharge and runoff from the industrial, municipal and non-industrial sources before release into the St. Clair River. A maximum concentration of 19 000 ng/L was detected in untreated leachate before carbon treatment. After carbon treatment, concentrations of 5 000 ng/L 1,4-dichlorobenzene were measured in the Township Ditch. The authors noted, however, that because samples were collected on one day only, their loading estimates cannot be assumed to reflect long-term average values.

In a 1987 monitoring program of wastewater treatment plants in the Vancouver area, 1,4-dichlorobenzene was detected in five of six treatment plant effluent samples from the Iona Island Wastewater Treatment Plant. The mean contaminant concentration for composite effluent samples was less than 240 ng/L (range <10 to 390 ng/L). The contaminant levels found in the two grab samples were 170 and 700 ng/L (Fanning et al., 1989). Since the 1987 data were collected, a deep sea outfall was put into service that replaces the previous surface discharge on Sturgeon Bank. The new outfall discharges primary treated effluent at depths of 70 to 100 metres into the Strait of Georgia, near Vancouver, British Columbia.

Loadings of contaminants in urban runoff from 12 urban centres in the Canadian Great lakes basin was reported by Marsalek and Schroeter (1988). Mean annual loadings of 1,4-dichlorobenzene into five sub-basins corresponding to Lakes Erie, Huron, Ontario, St. Clair and Superior ranged from 0.0003 to 0.013 kg/d with the highest loading measured from Lake Ontario: the combined basin loading value was 0.022 kg/d.

The effectiveness of three sewage treatment plants in Ontario (Welland, Waterloo and Galt) receiving municipal and industrial wastewater was reported by Melcer et al. (1988). Removal of 1,4-dichlorobenzene in the effluents over a 8 day period ranged from 53% in the Waterloo plant to 65% at the Galt plant (based on geometric means). Mean concentrations of 1,4-dichlorobenzene in the effluent ranged from 900 to 1 400 ng/L. The authors were not able to determine whether the removal was attributed to degradation or stripping as no air samples were taken. Removal of 1,4-dichlorobenzene from municipal wastewater at a pilot-scale plant and an aeration basin in southern Ontario was investigated by Melcer (1990). The mean removal of 1,4-dichlorobenzene from the pilot plant was 81.7% compared to 74.5% for the aeration basin. Stripping was the major removal mechanism whereas biodegradation and adsorption were limited removal mechanisms.

In a 1979 study of chlorobenzenes in bottom sediment, effluent and biota near a coking oven, a chlor-alkali chemical plant, a textile plant, and a control site in Nova Scotia, MacLaren Marex Ltd. (1979) reported levels of total 1,3- and 1,4-dichlorobenzene in water samples from the outfall into which the textile plant effluent was discharged and from a sewer manhole close to the plant. The two water samples contained levels of total 1,3- and 1,4-dichlorobenzene ranging from 7 500 to 9 400 ng/L.

Other Sources

Anaerobic transformation of the trichlorobenzenes (1,2,3-, 1,3,5-, 1,2,4-trichlorobenzenes) yielding dichlorobenzenes in laboratory columns packed with sediments from the Rhine river was reported by Bosma et al. (1988). After 300 days of continuous operation of the column at concentrations in the influent ranging from 300 to 500 nmol/L, the trichlorobenzenes were transformed into 1,2, 1,3- and 1,4-dichlorobenzenes. In experiments with individual isomers, 1,4-dichlorobenzene was formed from 1,2,4-trichlorobenzene. The fact that the sediment column containing 1,2,4-trichlorobenzene was transformed to 1,4-dichlorobenzene, indicates that specific enzymes and/or microorganisms are responsible for the observed transformations.

Three incinerator technologies were tested in the mid-1980's under Environment Canada's National Incinerator Testing and Evaluation Program (Environment Canada, 1985a; Environment Canada, 1988). The main objective of this program was to define the

optimum design and operating conditions to minimize emissions of concern. 1,4-Dichlorobenzene was not analyzed separately from the rest of the chlorinated benzenes, so there are no data which are relevant to this assessment.

Sources in the United States

Production of 1,4-dichlorobenzene in the United States has resulted in emissions from vents, exhaust fans, liquid product storage, handling and the vacuum system which services the vacuum stills (U.S. EPA, 1986). Emission factors, which are general estimates derived from site visit measurements, are presented in Table 3-4.

The majority of 1,4-dichlorobenzene produced in the United States is used in space, toilet bowl and refuse deodorants. These deodorant products contain up to 100% 1,4-dichlorobenzene. It is estimated that all 1,4-dichlorobenzene used in solid space and garbage deodorants will enter the atmosphere by sublimation during production or use (U.S. EPA, 1986).

An estimated 635 kg/yr of 1,4-dichlorobenzene were emitted into the atmosphere from pesticide formulation in 1978 (U.S. EPA, 1986). It is estimated that all 1,4-dichlorobenzene used in pesticide applications are released to the atmosphere; the exact amount of release depending on volatility and application form. There are no emissions data available for the production of polyphenylene sulfide (PPS), a commercial polymer with many industrial applications (U.S. EPA, 1986).

Analysis of 9 hazardous waste incinerators and 4 co-fired boilers in the United States suggested the emission of 1,4-dichlorobenzene as a product of incomplete combustion (PIC) (U.S. EPA, 1991). 1,4-Dichlorobenzene was emitted from full-scale boiler tests, with a geometric mean emission rate of 474.0 μg/min. 1,4-Dichlorobenzene was one of the most prevalent PICs from pilot-scale afterburner tests, emitted in at least 67% of the tests, with a geometric mean emission rate of 80.8 μg/min. 1,4-Dichlorobenzene exhibited significant emission rates from pilot-scale stack sampling; a mean emission rate of 255.8 μg/min was reported.

In 1987, about 15 kg of ashes were obtained from 18 municipal incinerators representing one-fourth of all the presently operating incinerators in the United States (Shane et al., 1990). Twenty ashes including fly ash, bottom ash and mixtures of the two were analyzed for a range of organic toxicants and mutagens. 1,4-Dichlorobenzene was detected in 3 ashes at concentrations ranging from not detectable to 51 μg/kg (detection limit - 5 ng/g).

The Toxic Release Inventory (TRI) database (Marcos, 1992), which monitors emissions from stationary industrial sources, showed a release of 602 tonnes for the year 1989 into air, water and land

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(Table 3-5) compartments. Of that amount, the majority of the emissions (85% or 514 tonnes) was released to the atmosphere. Comparison of environmental release to U.S. demand (36 300 tonnes) in 1990, indicates that 1.6% of the total demand of 1,4-dichlorobenzene was released to the environment.

4.0 ENVIRONMENTAL TRANSPORT, TRANSFORMATION AND LEVELS

Volatilization, photooxidation, partitioning to soil, sediment and biota, and aerobic degradation are the main processes affecting the transport and distribution of 1,4-dichlorobenzene in the environment.   Direct photolysis, hydrolysis and anaerobic degradation of 1,4-dichlorobenzene are not expected to play an important role in its environmental fate.

4.1 Transport and Distribution Between and Within Media

4.1.1 Air

Long range transport of 1,4-dichlorobenzene in the atmosphere is expected considering that the mean half-life in the atmosphere for photooxidation of 1,4-dichlorobenzene is approximately three weeks (Mackay et al., 1992).

1,4-Dichlorobenzene was detected in the rain-dissolved phase during seven rain events from February to April, 1984 in Portland, Oregon. The presence of 1,4-dichlorobenzene in the rain phase indicates that the substance persists long enough to be returned to the earth's surface by atmospheric washout (Ligocki et al., 1985).

The products of atmospheric reactions of 1,4-dichlorobenzene in the atmosphere are not well known. Irradiation of 5 ppm(v) 1,4-dichlorobenzene in the presence of 0.5 ppm NO_2 was conducted in ultrapure air for 20 hrs. Dichloronitrobenzene, dichlorophenol and dichloronitrophenol were the products identified (Spicer et al., 1985).

4.1.2 Water

Transport and distribution of a substance between and within media in the aquatic environment is dependent upon its solubility, movement of the water itself, exchanges at the air-water interfaces, adsorption to sediment and particulate matter, and bioconcentration in aquatic organisms. The residence time in water is also dependent upon the type of environmental conditions encountered, including temperatures, wind speeds, currents, and ice cover.

Fresh Surface Water

Laboratory and field studies indicate that 1,4-dichlorobenzene volatilizes readily from the water column to the atmosphere. At a concentration of 300 mg/L, 1,4-dichlorobenzene volatilized almost completely from aerated distilled water in less than 4 hrs and from non-aerated distilled water in less than three days (Garrison and Hill, 1972). Callahan et al. (1979) calculated a half-life from aerated distilled water of less than 30 minutes, compared to a half-life in unaerated water of less than 9 hrs. However, for the aerated solution, the authors did not recommend this value as aeration probably caused air-stripping of the 1,4-dichlorobenzene. Rippen et al. (1984) reported a volatilization half-life for 1,4-dichlorobenzene of 21 hours based on a OECD draft method for a 1-m water depth at 25°C (OECD, 1979). Schwarzenbach et al. (1979) reported that mass transfer to the atmosphere was the prevalent elimination mechanism for 1,4-dichlorobenzene from a lake. Using a simple steady-state model, the authors estimated that approximately 60 kg of 1,4-dichlorobenzene were transferred from Lake Zurich in Switzerland to the atmosphere. Based on physical/chemical parameters, Mackay et al. (1992) calculated a half-life in surface water of 672 to 4 320 hours (based on aqueous biodegradation half-life).

Volatilization was the major removal mechanism during studies carried out in an experimental marine mesocosm in Narragansett Bay, Rhode Island (Wakeham et al., 1983). The authors reported half-lives ranging from 10 days in summer to 13 and 18 days in winter and summer, respectively. Biodegradation and adsorption were found to be insignificant removal mechanisms. Comparing laboratory and field estimates, volatilization processes should be significantly faster in sea coastal waters, where the water body is usually more turbulent, than in the experimental tanks.

Zoeteman et al. (1979) estimated half-lives in surface water by observing the reduction in concentration of 1,4-dichlorobenzene at locations along the Rhine River in The Netherlands. Half-lives ranged from 1.1 to 26 days over a study period from 1972 to 1979.

Groundwater

Land disposal of sewage effluents to soil has been shown to provide a direct source for introduction of 1,4-dichlorobenzene into groundwater (Bouwer et al., 1981; 1984; Hutchins et al., 1983; 1984; Barber et al., 1988; Barber, 1988). Three mechanisms of importance in the transport and fate of 1,4-dichlorobenzene in soil percolation systems are biodecomposition, sorption and volatilization. The detection of 1,4-dichlorobenzene in groundwater after land application of wastewater in both Phoenix, Arizona (Bouwer et al., 1984) and Fort Devens, Massachusetts (Hutchins et al. 1983), indicated to the authors that although transformation of 1,4-dichlorobenzene may have occurred from percolation through the aerobic zone little degradation was observed in the anoxic aquifer. In Phoenix, concentrations of 1,4-

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dichlorobenzene were monitored in effluent and groundwater after application of secondary wastewater to infiltration basins during two sampling periods of three months each from 1980 to 1981 (Bouwer et al., 1984). Nonchlorinated secondary wastewater was used during the first sampling period, and chlorine was added to the effluent during the second sampling period. 1,4-Dichlorobenzene decreased during percolation through soil during the first sampling period with percentage concentrations decreases in the groundwater ranging from 33 to 48%. 1,4-Dichlorobenzene decreased to a lesser extent during the second sampling period with percentage concentration decreases from 3 to 24%. Hutchins et al. (1983) reported an average influent concentration of 0.243 μg/L and average effluent concentrations of 0.042 μg/L 1,4-dichlorobenzene, corresponding to a 83% removal for the study period from November 1980 to May 1981. The authors attributed the removal of 1,4-dichlorobenzene from soil columns by biodegradation and biosorption to be the predominant processes with abiotic degradation and volatilization as minor processes.

1,4-Dichlorobenzene was reported to be both persistent and mobile under anaerobic conditions in the leachate plume of a municipal landfill of North Bay, Ontario (Reinhard et al., 1984; Barker et al., 1986). 1,4-dichlorobenzene was detected 800 m from the landfill confirming the low sorption capacity of the North Bay sandy aquifer (0.01%). Attenuation of 1,4-dichlorobenzene appeared to be by dispersion only (Reinhard et al., 1984; Barker et al., 1986). The presence of 1,4-dichlorobenzene in a plume of contaminated groundwater at a distance of more than 3 500 m from sewage disposal beds near Falmouth, Massachusetts, in 1984, indicated little degradation during infiltration (Barber et al., 1988). The persistence of 1,4-dichlorobenzene in the aquifer (30 years) was attributed to redox conditions and nutrient limitations (carbon and oxygen) that were unfavourable for microbial degradation (Barber, 1988).

Wastewater

Greater than 99% removal of 1,4-dichlorobenzene was observed in activated-sludge treatment of wastewater in a California water pollution control plant (McCarty and Reinhard, 1980). An influent concentration of 1 900 ng/L was reduced to <20 ng/L, after waste segregation and activated sludge treatment. Hannah et al. (1986) evaluated pilot-scale less-than-secondary treatment processes designed to remove toxicants in Cincinnati, Ohio. A control activated sludge process was operated in parallel with five alternative treatment processes. The conventional activated sludge system provided the best removal of 1,4-dichlorobenzene and the facultative lagoon with its long detention time (25.6 days) were the most successful processes. The remaining four alternative processes were less effective in removing 1,4-dichlorobenzene, percent removals ranged from 19 to 65%.

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4.1.3 Soil

Volatilization from a soil surface can be an important transport mechanism, however, volatilization may be attenuated by sorption and leaching (Howard et al., 1990). Mackay and Yuen (1983) stated that the presence of organic or mineral sorbing suspended solids will reduce the volatilization rate by reducing the fugacity or partial pressure. Based on its estimated organic carbon sorption coefficient (K_{oc}) of 1 030 (calculated using the formula of Mackay et al. (1992)), 1,4-dichlorobenzene has a low soil mobility potential (McCall et al., 1981).

Evidence for leaching of 1,4-dichlorobenzene in soil was been reported by Wilson et al. (1981), and Elder et al. (1981). 1,4-Dichlorobenzene was applied at concentrations of approximately 0.2 mg/L to columns containing 140 cm packed sandy soil (0.09% organic carbon). Concentrations of 0.80 and 0.13 mg/L 1,4-dichlorobenzene applied to the column for 45 days resulted in a significant proportion of 1,4-dichlorobenzene percolating through the soil (37% and 49%, respectively). The remaining 63% and 51% were degraded or unaccounted for. The presence of total dichlorobenzenes (1,2-, 1,3- and 1,4-dichlorobenzene) in soil samples adjacent to hazardous waste disposal sites in Niagara Falls, New York, demonstrated that the compounds had leached from the dump sites. Bouwer et al. (1981) attributed the removal of 1,4-dichlorobenzene in secondary treated municipal wastewater applied to three soil columns to volatilization and biotransformation. A concentration decrease of 42% was reported over a two week period as water moved through the soil at the Phoenix, Arizona site.

Based on physical/chemical properties, Mackay et al. (1992) calculated a half-life in soil of 8 months, while Howard (1991) reported a half-life ranging from 4 weeks to 6 months.

4.1.4 Sediment

Based on physical/chemical properties and loadings of 1,4-dichlorobenzene from the Niagara River to Lake Ontario, Oliver and Charlton (1984) estimated that only a small percentage of 1,4-dichlorobenzene (1%) is expected to sorb to sediments. Releases from sediments can occur by resuspension, redistribution and desorption (Oliver, 1985; Oliver and Charlton, 1984). Little anaerobic degradation of 1,4-dichlorobenzene is expected to occur below surface level. Once in the sediments, 1,4-dichlorobenzene is expected to equilibrate between the pore water or the organic phase (Di Toro et al., 1991).

Surveys of Lake Superior, Huron, Erie, and Ontario indicated that 1,4-dichlorobenzene accumulated in bottom sediments, with high concentrations at the 0 to 5 cm core depth and decreasing dramatically from the 5 to 8 cm core depth (Oliver and Nicol,

1982). There was no significant change in the relative proportions of 1,4-dichlorobenzene when proceeding down Lake Ontario cores, indicating that little degradation had occurred. Oliver and Charlton (1984) observed that 1,4-dichlorobenzene adsorbed onto bottom sediments from the Niagara River and were resuspended into the overlying water and redistributed in sedimentation basins in Lake Ontario. The authors were able to estimate the approximate amount of 1,4-dichlorobenzene loading to the Niagara River (23 kg/day), to the lake sediments (0.3 kg/day) and the downflux of contaminant as percentage of loading (1%) in 1982. Oliver and Charlton (1984) also suggested that due to its vapour pressure, volatilization would be a major removal mechanism from Lake Ontario.

Based on field observations, Oliver (1985) reported that a 1 mm thick sediment layer in three sedimentation basins in Lake Ontario is continually being resuspended then resettling to the bottom. With a near bottom temperature of 4 °C and the active layer containing about 2% solids, the average desorption half-life under these conditions is about 60 days. Oliver (1985) observed a slow desorption of 1,4-dichlorobenzene in a laboratory study using three sediment samples representing the three basins from Lake Ontario. After 39 days at 4°C (average near bottom temperature in Lake Ontario), 30% of 1,4-dichlorobenzene was desorbed from sediments during continuous purging experiments. Increasing the temperature to 20°C and 40°C after 39 days, resulted in the desorption rate increasing to 39% and 60%, respectively. The contribution of 1,4-dichlorobenzene loadings via desorption from bottom sediments in Lake Ontario was estimated to be minimal (11 kg/yr) in comparison with the loading estimates from the Niagara River (5 800 kg/yr) to Lake Ontario (Oliver, 1985). Based on a mass balance calculated by Schwarzenbach et al. (1979), a maximum of 2 kg of the 60 kg of 1,4-dichlorobenzene eliminated during one year was incorporated into bottom sediments.

Based on a review of the literature, Mackay et al. (1992) selected a half-life of approximately 2 years for 1,4-dichlorobenzene in the first 1 cm of sediment.

Oliver and Nicol (1982) compared the relative proportions of different chlorobenzene congeners in surface and buried (older) sediments, and concluded that there was little evidence of either microbial oxidation or anaerobic dehalogenation of chlorobenzenes in sediments from Lake Ontario.

4.1.5   Biota

1,4-Dichlorobenzene has the potential to be available to organisms in the water column and sediments. However, due to its water solubility and vapour pressure, Oliver and Charlton (1984) suggested that 1,4-dichlorobenzene is more likely to remain dissolved or volatilize from water. Oliver (1987b) observed that

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the potential for bioaccumulation for benthic organisms comes mainly from the sediment pore water rather than from ingestion of contaminated sediment particles.

Bioconcentration factors for a number of organisms have been determined for 1,4-dichlorobenzene[1]. Barrows et al. (1980) and Veith et al. (1980) reported a bioconcentration factor (BCFs) in bluegill sunfish (Lepomis macrochirus) of log 1.78 (60) (whole fish basis). Bluegill sunfish were maintained at equilibrium in a flow-through system for 14 days (mean water concentration: 7.89 μg/L) and exposed to pollutant-free water for a period of 7 days. Half-lives in tissue of less than one day were estimated by both authors. The Aquatic Toxicity Research Group (ATRG, 1987) exposed juvenile male flagfish (4 to 6 months old) for a 28-day period to unmeasured concentrations of 5 μg/L followed by a 14-day depuration period. Measured bioconcentration factors for 1,4-dichlorobenzene were 296 for whole fish and 3 590 for lipids. A reported half-life of <1 day indicates that 1,4-dichlorobenzene is rapidly depurated. Oliver and Niimi (1983) reported slightly higher BCFs of log 2.56 and 2.85 (370 and 720, respectively) for rainbow trout (Oncorhynchus mykiss) at low and high exposure levels (whole fish basis), respectively. Equilibrium was attained within seven to eight days over the 105 and 119 day study, respectively. Oliver (1987b) studied the uptake and depuration of 1,4-dichlorobenzene in spiked Lake Ontario sediments (4.6% organic carbon) by oligochaete worms. A half-life of less than 5 days was reported in the worms after a 79 day exposure period in the contaminated sediment and a 84 day depuration period in clean Lake Superior sediment. Based on physical/chemical properties, Mackay et al. (1992) calculated a half-life in biota of <1 day in fish, and <5 days in worms.

Pereira et al. (1988) determined BCFs for four species of biota in a field study in the vicinity of an industrial outfall in the Calcasieu River estuary, Louisiana. BCFs (lipid basis) for the spotted sea trout (Cynascion nebulosis), blue catfish (Ichtalurus furcatus), Atlantic croakers (Micropogonias undulatus) and blue crabs (Callinectes sapidus), were log 4.09 (12 302), log 3.51 (3

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[1] A bioconcentration factor is defined as:

$$BCF = \frac{\text{Concentration of chemical at steady state in organism (wet weight)}}{\text{Mean concentration of chemical in water}}$$

or

$$BCF = \frac{\text{Uptake rate constant at steady state}}{\text{Elimination rate constant}}$$

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226), log 3.91 (8 128) and log 4.53 (33 884), respectively.

Calamari et al. (1982) reported bioconcentration factors resulting from tests on developmental stages of rainbow trout (Oncorhynchus mykiss). In 7 and 60-day bioconcentration tests on developmental stages from egg to alevins in a continuous flow system, bioconcentration factors ranged from 100 to 450 and increased to 1 400 at day 23 of the 60 day early life stage study in the rainbow trout. Ahmad et al. (1984) reported a comparable bioconcentration factor of Log 2.05 (112) (whole fish) in fathead minnows (Pimephales promelas) exposed for 32 days during an early life-stage toxicity test.

4.2 Transformation

4.2.1 Biodegradation

Aerobic degradation of 1,4-dichlorobenzene has been observed in wastewater, groundwater, soil, and in sediment under acclimated conditions. However, poor biodegradation activity was reported under anaerobic conditions. Reductive dechlorination of higher chlorobenzenes to 1,4-dichlorobenzene was reported in anaerobic sediment.

Air

Howard (1989) reports that 1,4-dichlorobenzene will exist predominantly in the vapour phase in the atmosphere. The half-life for the vapour phase reaction of 1,4-dichlorobenzene with photochemically produced hydroxyl radicals in the atmosphere is estimated by Howard (1989) to be 31 days.

Surface water

The persistence of 1,4-dichlorobenzene under denitrification conditions and at concentrations commonly found in surface and groundwater was demonstrated by Bouwer and McCarty (1983). No biotransformation of 1,4-dichlorobenzene was reported after 11 weeks of incubation, in the presence of denitrifying bacteria and at initial concentrations ranging from 41 to 114 μg/L.

Wastewater

Microbial degradation of 1,4-dichlorobenzene under the static-flask-culture test was reported by Tabak et al. (1981) using wastewater inoculum, a seven-day acclimation period, and three weekly subcultures. After seven days, 55% 1,4-dichlorobenzene added at 5 mg/L was degraded and 37% was degraded at the 10 mg/L levels. A slight increase in biodegradation was observed after the first subculture, however, subsequent subcultures demonstrated a reduction of the biodegradation rate. During the third subculture, at 10 mg/L of substrate, no biodegradation was observed. This

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result was ascribed either to loss of the synergistic activities of the original microbial population or to the build-up of toxic metabolites (Tabak et al., 1981).

1,4-Dichlorobenzene was biotransformed during long-term (3 years) continuous-flow aerobic column studies with acetate as the primary substrate (Bouwer and McCarty, 1985). Average removal efficiencies of 1,4-dichlorobenzene in a 1 mg/L acetate-grown biofilm column, ranged between 98% (after 9 months removal) and 99% (between 9 and 36 months removal) and after an acclimation period of 20 days. An increased acclimation period (60 days) was required in 5 mg/L biofilms with removal efficiencies at 84% (after 5 months) and 98% (between 5 and 32 months). In similar studies with mixed bacterial cultures and under aerobic and methanogenic conditions, significant degradation (98-99%) was observed in aerobic biofilm columns (Bouwer and McCarty, 1984; Bouwer, 1985). No transformation (0%) was reported in the methanogenic biofilm columns (Bouwer, 1985).

Canton et al. (1985) investigated the biodegradability of 1,4-dichlorobenzene in a Repetitive Die Away Test (RDA) under unacclimated conditions. The test was not endorsed by OECD as the compounds volatilized (approx. 43% evaporated from nonaerated standardized medium within two days at room temperature). 1,4-Dichlorobenzene was biodegraded without adaptation of the inoculum, with an unacclimated biodegradation half-life of 2 to 3 weeks.

Groundwater

Aquifer columns simulating saturated-flow conditions typical of a river water/groundwater infiltration system, indicated that 1,4-dichlorobenzene was biodegraded under aerobic conditions (Kuhn et al., 1985). After three weeks, the overall level in the effluent was reduced by 90% of the input concentration (0.2 μM). 1,4-Dichlorobenzene appeared to be utilized by bacteria as the sole energy and carbon source. Column studies conducted under anaerobic conditions showed no biotransformation of 1,4-dichlorobenzene over a 90 day period.

Based on physical-chemical parameters, Mackay et al. (1992) calculated a half-life in groundwater of 1 344 to 8 640 hours (based on an aqueous aerobic biodegradation half-life).

Soil

Schraa et al. (1986) reported degradation of 1,4-dichlorobenzene by the aerobic bacteria Alcaligenes strain A175 isolated from a mixture of soil and water samples. Doubling time of the organism in batch culture was about 8 hours, and degradation of 1,4-dichlorobenzene was observed after about 2 months. Growth of 1,4-dichlorobenzene on Alcaligenes strain OBB65, a gram negative, peritrichously flagellated rod, was demonstrated by de

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Bont et al. (1986). The doubling time of the organism on 1,4-dichlorobenzene was reported to be 15 hours. No transformation of 1,4-dichlorobenzene was observed for a period of 500 days in an aerobic soil percolation column inoculated with cells of Pseudomonas testosteroni and containing a mixture of 1,2-, 1,3-, and 1,4-dichlorobenzene (Schraa and van der Meer, 1987). Haider et al. (1981) demonstrated that a dichlorobenzene mixture (60% 1,4-dichlorobenzene and 40% 1,2-dichlorobenzene) was slowly degraded by the soil bacteria (Pseudomonas sp.) cultivated on benzene (up to 12% release of marked CO_2 after 7 days). These compounds proved to be resistance to biodegradation in a soil grab sample test (7% CO_2 releases after 10 weeks). It was suggested that a longer adaptation period was required, or that degradation products were harmful to the microorganisms (Haider et al., 1981). van der Meer et al. (1987) tested the ability of native soil microbial from Rhine sediments to aerobically biodegrade 1,4-dichlorobenzene in a mixture of sand and sediment from the Rhine River. No biodegradation of 1,4-dichlorobenzene by the native soil microbial populations was observed during the 200 day experiment. Upon inoculation of cells of Pseudomonas sp. strain P51 with influent concentrations of 600 μg/L 1,4-dichlorobenzene, rapid degradation of 1,4-dichlorobenzene was observed over a 77 day period until a threshold concentration of 40 μg/L was reached. In two additional inoculation experiments with soil columns containing influent concentrations of 80 and 10 μg/L 1,4-dichlorobenzene, no degradation was observed for 1,4-dichlorobenzene. A threshold concentration of 30 μg/L was observed for aerobic degradation of 1,4-dichlorobenzene. The authors suggested that the threshold concentration below which no degradation of 1,4-dichlorobenzene occurred due to a complex number of interactions between soil adsorption sites, metabolic characteristics of the bacteria and the substrates present.

The potential for biodegradation of 1,4-dichlorobenzene at low concentrations was demonstrated by Bouwer and McCarty (1983; 1984, 1985) using the biofilm kinetic approach. In laboratory studies simulating aquifer conditions, mineralization of 1,4-dichlorobenzene was observed in the presence of acetate as a primary substrate. Ninety-eight percent of 1,4-dichlorobenzene was removed after a 10 day acclimation period and 10.8 μg/L of influent solution. However, no biotransformation of 1,4-dichlorobenzene was reported under denitrification conditions in batch studies after 11 weeks of incubation and at concentrations of 41 to 114 μg/L (Bouwer and McCarty, 1983).

Biodegradation of alpha-hexachlorocyclohexane (alpha-HCH) in a contaminated soil slurry (organic carbon content, 0.9± 0.06%) seeded with anaerobic sewage sludge resulted in an increase in 1,4-dichlorobenzene (Bachmann et al., 1988). 1,4-Dichlorobenzene was present in the original soil slurry at a concentration of 1.53 mg/kg and increased to approximately 3.5 mg/kg after 100 days. A

degradation pathway of alpha-HCH proposed by the authors involved conversion of alpha-HCH to pentachlorocyclohexene (PCCH) by dehydrochlorination, followed by conversion of PCCH to tetrachlorobenzene and trichlorobenzene, which in turn were further dechlorinated to 1,2- and 1,4-dichlorobenzene.

Sediment

Holliger et al. (1992) reported the reductive dechlorination of 1,3,5-trichlorobenzene to 1,4-dichlorobenzene in enrichment cultures supplemented with lactate as the electron donor and inoculated with anaerobic sediment from the Rhine River. 1,4-Dichlorobenzene was formed after a lag phase of approximately one year. Bosma et al. (1988) reported the anaerobic biotransformation of 1,2,4-trichlorobenzene to 1,4-dichlorobenzene after 300 days in a laboratory column packed with Rhine river sediment. 1,2,4-Trichlorobenzene was biotransformed to 1,4-dichlorobenzene as an intermediate and monochlorobenzene and chloride as end products, following a lag of 7 days. However, 1,4-dichlorobenzene was transformed only after a complete removal of 1,2-dichlorobenzene and a relatively high concentration in the influent solution (300-500 nmol/L).

Sewage Sludge

Removal of 1,4-dichlorobenzene from sewage sludge by aerobic biodegradation, volatilization and stripping has been reported. 1,4-Dichlorobenzene has also been reported to be a byproduct of the dechlorination of higher chlorinated benzenes in sewage sludge.

Calamari et al. (1982) reported slow biodegradation of 1,4-dichlorobenzene in treatment plant sludge. Degradation began after 14 days at 20°C and complete disappearance was observed at the end of the 28-day incubation period. Degradation was dependent on the concentration of the compound; complete degradation of 1,4-dichlorobenzene was observed at low concentrations (8 mg/L), compared to 38 and 0% degradation in two solutions containing high concentrations of 1,4-dichlorobenzene (40 mg/L).

Biodegradation of 1,4-dichlorobenzene by a _Pseudomonas_ species isolated from activated sludge was reported by Spain and Nishino (1987). With 1,4-dichlorobenzene as the sole carbon source, biodegradation began after 10 months of enrichment and the doubling time for the _Pseudomonas_ species was 5 hours. The fate of sewage sludge spiked with 1,4-dichlorobenzene, with and without azide to prevent biological activity, was reported by Kirk et al. (1989). The authors attributed the removal of 80% of 1,4-dichlorobenzene during the 32 day period to biodegradation. Anaerobic digestion was not considered a significant removal in azide treated sludge.

The biodegradability of 1,4-dichlorobenzene was studied using the Organization for Economic Cooperation and Development (OECD)

closed bottle test (Topping, 1987). The major proportion of 1,4-dichlorobenzene spiked into raw sewage of an activated sludge plant (>67%) was found in the supernatant liquor after 2 hours sedimentation, 10% volatilized, and 22% was adsorbed onto primary settled solids. At a high aeration rate (2 L/min), volatilization was the major removal mechanism for 1,4-dichlorobenzene. In a modified porous pot test, operated at a low aeration rate, at temperatures of 8, 15, 20°C, and a sludge retention time of 3 and 6 days, overall removal was high (>95%). Under normal operating conditions, >76% is expected to be removed by biodegradation processes, with the overall removal greater than 95% during aeration of activated sludges.

The fate of contaminated sludge applied to soil at two sample depths (0 cm to 15.2 cm and 22.9 cm to 38.1 cm) over a 241 day period was studied by Demirjian et al. (1987). Sludge containing 415 μg/kg 1,4-dichlorobenzene was applied to a field plots at an application rate of 448144 kg/ha. After 5 days, the concentration of 1,4-dichlorobenzene increased to 432 μg/kg at the 1 cm to 15.2 cm depth and was not detected at the lower depth. After 122 days, 1,4-dichlorobenzene had degraded to 104 μg/kg at the upper depth and was detected at 41 μg/kg at the 22.9 to 38.1 cm depth, indicating possible leaching. 1,4-dichlorobenzene had increased to 230 μg/kg in upper cores after 241 days and decreased to 28 μg/kg in the lower core. The authors attributed losses of 1,4-dichlorobenzene in the surface layer of the soil to degradation or adsorption.

Fathepure et al. (1988) reported the reductive dechlorination of hexachlorobenzene to tri- and dichlorobenzenes in anaerobic sewage sludge. Greater than 90% of hexachlorobenzene was degraded to 1,3,5-trichlorobenzene, while the remainder was transformed to 1,2,4-trichlorobenzene and ultimately 1,4-dichlorobenzene.

4.2.2   **Abiotic Degradation**

Hydrolysis

Under the range of conditions found in the Canadian environment and the difficulty with which aryl halides undergo nucleophilic substitution, hydrolysis of 1,4-dichlorobenzene in ambient water is unlikely to occur (Morrison and Boyd, 1987; Callahan et al., 1979; Oliver et al., 1979). The half-life due to hydrolysis was estimated to be greater than 900 years in groundwater beneath land disposal sites (Ellington et al., 1988).

Photolysis

Because 1,4-dichlorobenzene absorbs light only below 300 nm, photolysis in water is not believed to be a significant transformation process (Howard et al., 1990). However, Mansour et al. (1989) reported photodegradation of 1,4-dichlorobenzene in

distilled water and fresh water under sunlight irradiation. A half-life of 25 days was estimated following direct photolysis of 1,4-dichlorobenzene in lake water from April to May, 1986. The photolytic half-life of 1,4-dichlorobenzene in distilled water and river water under laboratory conditions ranged from 22.8 to 17.5 hours, respectively. The decomposition of 1,4-dichlorobenzene has been reported in the laboratory with a half-life of approximately 5 minutes (Oliver et al., 1979). 1,4-Dichlorobenzene was irradiated at 300 nm in a photoreactor in the presence of TiO_2 (anatase phase), a common constituent of clays, sediments and soils. However, when aqueous solutions of 1,4-dichlorobenzene were irradiated as 1.0% slurries with naturally occurring clays and suspended sediments collected from several rivers flowing into the Great Lakes, no decomposition was observed for irradiation periods up to 4 hours (Oliver and Carey, 1986; Oliver et al., 1979). The suspended sediments acted as light absorbents and reduced the rate of photolysis by shielding the pollutant from the available light.

Photooxidation

The primary mechanism for 1,4-dichlorobenzene transformation in the atmosphere is through reaction with hydroxyl radicals (OH) (U.S. EPA, 1989a). 1,4-Dichlorobenzene will react with photochemically produced hydroxyl radicals in the atmosphere with an estimated photooxidation half-life ranging from 3 to 31 days (Callahan et al., 1979; Rippen et al., 1984; Howard et al., 1989). Mackay et al. (1992) estimated a mean half-life in the atmosphere for photooxidation and advection processes of approximately three weeks. Based on physical-chemical parameters, Mackay et al. (1992) calculated a half-life in air of 201 to 2 006 hours (based on photooxidation in air). Based on rate constants and hydroxyl radical concentrations in the range of 3×10^6 to 3×10^5 $cm^3molecule^{-1}sec^{-1}$ for a fairly polluted and pristine environment, the atmospheric half-lives of 1,4-dichlorobenzene were within the range of 8 to 84 days (U.S. EPA, 1989a). Singh et al. (1983) reported a 2.6% loss of 1,4-dichlorobenzene involving the hydroxy radical in one 12 hour sunlit day. A residence time (the time for removal of all of the chemical initially present in the atmosphere) of 38.6 days was calculated by Singh et al. (1981) based on an estimated average daily (24-h) OH abundance of $10^6molecule$ cm^3. Dann (1985, unpubl.) estimated a longer residence time of 165 days based on a daily (24h) average hydroxyl radical abundance of 10^6 mol/cc in a typical urban environment in summer and a daily rate loss (12 h sunlit day) of 0.6%. The experimentally derived rate constant for the vapour-phase reaction between 1,4-dichlorobenzene and photochemically produced hydroxyl radicals was reported to be 0.32 $\times 10^{-12}$ $cm^3molecule^{-1}sec^{-1}$ (Atkinson et al., 1985).

4.3   Environmental Concentrations

4.3.1   Air

The mean concentration of 1,4-dichlorobenzene measured in extensive (24 hour) samples of ambient air from 23 urban sites from five provinces throughout Canada during the period October, 1988 through December 1990 was 0.92 $\mu g/m^3$ (Environment Canada, 1991, unpubl.) (Table 4-1).   Median concentrations for urban and industrial sites ranged from 0.17 to 2.06 $\mu g/m^3$. Daily maximum concentrations were highest in industrial parts of Toronto (maximum 15.7 $\mu g/m^3$), Windsor (maximum 14.6 $\mu g/m^3$), and Montreal (maximum 6.45 $\mu g/m^3$).   The fact that 99% of the samples were above the detection limit (0.05 $\mu g/m^3$) indicates that this substance is ubiquitous.

Rain and air sampling for seven rain events in Portland, Oregon in 1984 was reported by Ligocki et al. (1985).   Mean concentrations of 4.8 ng/L 1,4-dichlorobenzene in dissolved rain and 120 ng/m^3 in the gas phase were measured. Pankow et al. (1984b) sampled four rain events in spring and five rain events in winter, 1982 in Portland, Oregon. Mean dissolved rain concentrations for 1,4-dichlorobenzene ranged from 0.66 ng/L during spring and 5.5 ng/L during winter.

4.3.2   Water

Fresh Surface Water

1,4-Dichlorobenzene was measured in the Great Lakes and associated rivers in Canada between 1980 and 1989 at concentrations ranging from 0.19 to 310 ng/L (Table 4-2). Concentrations of 1,4-dichlorobenzene were at their peak in the early 1980s and have decreased in recent years. Oliver and Nicol (1982) sampled the open waters of Lakes Huron and Ontario, and the Grand and Niagara Rivers for chlorobenzene concentrations between April and November 1980. The authors reported mean concentrations of 45 ng/L and 4 ng/L 1,4-dichlorobenzene for Lakes Ontario and Huron, respectively. Concentrations of 1,4-Dichlorobenzene ranged from a mean of 10 ng/L in the Grand River and to between 1 and 94 ng/L in the Niagara River.   1,4-Dichlorobenzene was measured in 1981 in the Niagara River water near Niagara-on-the-Lake (NOTL) and Fort Erie. Concentrations of 1,4-dichlorobenzene were higher at NOTL (10.5 ng/L) compared to 1.0 ng/L at Fort Erie (Fox et al., 1983). A comparison of the concentrations of 1,4-dichlorobenzene at three sites along the Niagara River, the start of the river (Fort Erie) (1.7 ng/L), below a chemical dumpsite and a chemical manufacturing discharge just above Niagara Falls, New York (94 ng/L), and at the mouth of the river (Niagara-on-the-Lake) (29 ng/L), was reported by Oliver (1987a).   Oliver and Nicol (1984) reported a maximum concentration of 310 ng/L, and a mean of 36 ng/L 1,4-

dichlorobenzene whole water samples collected weekly over a two year period (1981 to 1983) from the Niagara River at Niagara-on-the-Lake (NOTL), Ontario. A mean concentration of 1.6 ng/L and maximum level of 3.8 ng/L 1,4-dichlorobenzene was reported by Stevens and Neilson (1989) in samples of whole water in Lake Ontario in 1986. Mean concentrations for Lakes Erie, Huron and Superior were 1.15, 0.29 and 0.34 ng/L 1,4-dichlorobenzene, respectively. 1,4-Dichlorobenzene was detected in ambient water collected at the head (Fort Erie) and mouth (Niagara-on-the-Lake) of the Niagara River for the period 1986 to 1987 and 1988 to 1989 (The Niagara River Data Interpretation Group (NRDIG), 1988; 1990). Mean concentrations ranged from 0.82 (Fort Erie) to 2.47 ng/L (NOTL) and 0.97 (Fort Erie) to 1.93 ng/L (NOTL), for 1986 to 1987 and 1988 to 1989 data, respectively.

Levels of 1,4-dichlorobenzene along the St. Clair, Ontario in 1985 were comparable to those in the Niagara River in recent years. Forty-litre water samples were collected at locations upstream and downstream of a petrochemical industrial site at Sarnia along the St. Clair River between August and October of 1985 (Chan and Kohli, 1987). Levels of 1,4-dichlorobenzene ranged from 0.29 to 1.7 ng/L.

Based on a monitoring program of surface and raw drinking water supplies in six provinces (Alberta, Quebec, New Brunswick, Nova Scotia, Newfoundland, and Prince Edward Island) conducted from February 1987 to May 1989, the levels of 1,4-dichlorobenzene in surface water ranged between 0.57 to 130 ng/L in the provinces where it was detected (Environment Canada, 1992).

Marine Water and Estuarine Water

No information was found in the literature on 1,4-dichlorobenzene levels in marine or estuarine water in Canada.

Industrial and Municipal Wastewater

Concentrations of 1,4-dichlorobenzene in industrial and municipal wastewater are presented in Table 3-3 and discussed in Section 3.2.5.

Runoff

Marsalek and Schroeter (1988) reported concentrations of 1,4-dichlorobenzene in runoff from 12 urban centres in the Canadian Great Lakes basin (Lakes Erie, Huron, Ontario, St. Clair and Superior). A mean concentration of 89 ng/L 1,4-dichlorobenzene was measured in the stormwater (Table 4-2).

Ground Water

In Canada, elevated levels of 1,4-dichlorobenzene have been detected in ground water adjacent to waste disposal and landfill

sites (Table 4-4). 1,4-Dichlorobenzene was detected in groundwater at a former disposal site for waste oils and solvents near Ville Mercier, Quebec (Martel and Ayotte, 1989). Analysis of groundwater from the Ville Mercier area for the period 1988 to 1989 revealed concentrations of 1,4-dichlorobenzene ranging from 1 100 to 3 800 ng/L in samples in contact with contaminated oil (detection limit 50 ng/L). Characterization studies of a chemical production site near Napierville, Quebec in 1987 and 1988 were conducted to determine the extent of contamination of chlorobenzenes and naphthalene in groundwater (Foratek International Inc., 1987; Foratek Inc., 1989). Highest levels of 1,4-dichlorobenzene were found near the warehouse at the plant site. Concentrations of 1,4-dichlorobenzene in groundwater at the site ranged from non-detectable (<8 μg/L) to 23 100 μg/l and up to 63 800 μg/L in the oil phase. This site is targeted for remediation.

1,4-Dichlorobenzene was detected in groundwater near a municipal landfill near North Bay, Ontario in 1981 (Reinhard et al., 1984). Concentrations of 1,4-dichlorobenzene were found in groundwater beneath the landfill (40 000 ng/L) and 800 m away (7 000 ng/L) where the leachate-contaminated groundwater emerged. The detection of 1,4-dichlorobenzene 800 m from the source confirmed the persistence of this compound. Pankow et al. (1984b) reported the presence of 1,4-dichlorobenzene in groundwater downgradient of a landfill in Burlington, Ontario, between February and March, 1983. Concentrations in the groundwater ranged from non-detectable (11 ng/L) to 2 000 ng/L with an average concentration of 880 ng/L. In a later study (October to November, 1983), a mean concentration of 170 ng/L 1,4-dichlorobenzene was reported in groundwater at the Burlington, Ontario landfill (Pankow et al., 1985).

Zoeteman et al. (1980) reported a concentration of 3 000 ng/L 1,4-dichlorobenzene in groundwater in The Netherlands.

4.3.3   Soil

Characterization studies of a chemical production site near Napierville, Quebec in 1987, 1989 and 1990 were conducted to determine the extent of contamination of chlorobenzenes and naphthalene in soil by accidental discharge in 1968 (Foratek International Inc., 1987; Foratek Inc., 1989; ADS associés Itée, 1990) (Table 4-5). A total of 28 samples were taken to determine the limits of the contaminated soil. Highest levels of 1,4-dichlorobenzene were found near the warehouse and the railway tracks at the plant site (Table 4-5). Levels of 1,4-dichlorobenzene ranged from non-detectable (<2 μg/g) to 2 460 μg/g in soil samples. A study of the nature and extent of soil contamination in a drainage ditch near the plant was carried out in 1990 (ADS associés Itée, 1990). Levels of 1,4-dichlorobenzene measured in the soil ranged from non-detectable (<0.1 μg/g) to 4.3 μg/g. The site is targeted for remediation.

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4.3.4   Bottom Sediment

Levels of 1,4-dichlorobenzene in bottom sediments in Canada are restricted mainly to concentrations in samples collected in the late 1970s to mid 1980s, with the exception of a recent study of bottom sediments carried out in 1992 in British Columbia (Table 4-5). 1,4-Dichlorobenzene levels were reported in bottom sediments in the Niagara and St. Clair Rivers, Lake Ontario, and in the East River in Nova Scotia.

Concentrations of 1,4-dichlorobenzene were reported in sediment near a municipal sewage outfall at Victoria, British Columbia (EVS Consultants, 1992). Municipal wastes from Victoria and the surrounding area are disposed of in the nearshore environment at Macaulay Point and Clover Point. Studies were conducted at the mouth of each outfall, to the east and west of the outfall and at distances of 100, 400, 800 and up to 1600 m from the outfall discharge. Elevated concentrations of 1,4-dichlorobenzene were present in the sediments at the outfall terminus and at the 100 m stations. Levels of 1,4-dichlorobenzene ranged from 1 to 1 710 ng/g (dry weight) (mean 141 ng/g) at the Macaulay Point outfall (Table 4-5).

Oliver and Pugsley (1986) detected concentrations of 1,4-dichlorobenzene in bottom sediments along the St. Clair River in the vicinity of Sarnia, Ontario in 1985. Maximum concentrations of total dichlorobenzene isomers (1,2-, 1,3- and 1,4-dichlorobenzene) were reported at 31 000 ng/g (average 5 243 ng/g) in the sediment between 1984 and 1985. Based on levels reported for the dichlorobenzene isomers in the same area by Oliver (pers. comm., 1992), it was estimated that the relative composition of 1,3-dichlorobenzene, 1,2-dichlorobenzene, and 1,4-dichlorobenzene was approximately 58, 32 and 10%, respectively of the total. Therefore, using the average value of 5 500 ng/g, it was estimated that the concentration of 1,4-dichlorobenzene in the St. Clair River sediments was 524 ng/g.

Examination of the vertical distribution of 1,4-dichlorobenzene in bottom sediments in Lake Ontario in 1981 near the mouth of the Niagara River, provides insight into the history of chlorobenzene production and loading to the Niagara River. U.S. production of chlorobenzenes was greatest from 1960 to 1970 as is evident from cores at the 7 to 10 cm depth (1 100 to 880 ng/g) (Durham and Oliver, 1983). Concentrations of 1,4-dichlorobenzene at the 0 to 3 cm depth ranged from 110 to 270 ng/g. Samples containing 1,4-dichlorobenzene were collected from the top 2 cm of sediment from five sites in western Lake Ontario from April to July 1981 (Fox et al., 1983). Concentrations ranged from 250 to 1 300 ng/g in April to 33 to 112 ng/g in July. Oliver and Nicol (1982) reported that mean concentrations of 1,4-dichlorobenzene in bottom sediment from three Great Lakes locations in 1980 were highest in Lake Ontario (94 ng/g) followed by Lake Huron (16 ng/g), Lake Erie

(9 ng/g) and Lake Superior (5 ng/g). The concentrations of 1,4-dichlorobenzene in Lake Ontario in the vicinity of the Niagara River was reported by Oliver and Charlton (1984). Levels in surficial sediment ranged from 110 to 150 ng/g (dry weight) 1,4-dichlorobenzene, with a mean concentration of 63 ng/g. In 1981, ten stations, sampled for concentration and distribution of 1,4-dichlorobenzene in bottom sediments along the St. Lawrence River were lower than levels measured in the Great Lakes (Merriman, 1987). Concentrations of 1,4-dichlorobenzene in bottom sediments ranged from <1 near Kingston, Ontario, Cape Vincent, New York and Morrisburg, Ontario to a maximum of 9 ng/g near Cornwall, Ontario. The presence of 1,4-dichlorobenzene in sediment in the North Saskatchewan River was reported by Ongley et al. (1988) in 1983. Bottom sediment consisting primarily of sand-sized material contained levels of 1,4-dichlorobenzene ranging from non-detectable to 107 ng/g (detection limit not specified).

In a 1979 study of chlorobenzenes in bottom sediment near a coking oven, a chlor-alkali chemical plant, a textile plants, and a control site in Nova Scotia, MacLaren Marex Ltd. (1979) reported levels of total 1,3- and 1,4-dichlorobenzene in bottom sediment at all four locations. Highest levels were found in sediments from the chlor-alkali plant (37 to 1 000 ng/g), whereas the remaining sites showed comparable results: the textile plant (92 to 440 ng/g), and the control site (150 to 620 ng/g).

4.3.5   Suspended Sediments

In suspended sediments in Canada, highest concentrations were reported in the Niagara River in the early to mid 1980s (Table 4-5). Niagara River water samples collected between August 1985 and August 1986 were analyzed for 1,4-dichlorobenzene in suspended solids under acidic and basic conditions (Maguire et al., 1989). 1,4-Dichlorobenzene was detected in suspended solids at a concentration of 28 ng/L. Fox et al. (1983) measured 1,4-dichlorobenzene in suspended sediments at two sites along the Niagara River at Niagara-on-the-Lake (NOTL) and Fort Erie, between June and September of 1981. Concentrations of 1,4-dichlorobenzene ranged from 96 ng/g (dry weight) in September to 680 ng/g in July at NOTL. 1,4-Dichlorobenzene was not detected in Fort Erie sediment, indicating sources of this contaminant along the Niagara River. 1,4-Dichlorobenzene concentrations measured in suspended sediments in earlier studies at Niagara-on-the-Lake support the levels reported by Fox et al. (1983). Kuntz and Wary (1983) reported a mean value of 180 ng/g (dry weight) in suspended sediments in the lower Niagara River from April 1979 to December 1981. Oliver and Charlton (1984) collected settling particles from sediment traps suspended at 20 to 68 m depth in the water column from September to October, 1982 in the Niagara Basin of Lake Ontario. Concentrations of 1,4-dichlorobenzene in the sediment traps ranged from non-detectable to 160 ng/g (dry weight) (detection limit not specified).

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In 1981, ten stations were sampled for the concentration and distribution of contaminants in suspended sediments along the St. Lawrence River (Merriman, 1987). Concentrations of 1,4-dichlorobenzene were consistent along the river, ranging from 17 ng/g at Brockville, Ontario and Alexandria, New York to 27 ng/g at Morrisburg, Ontario. Between 1981 and 1987, Germain and Langlois (1988) collected dissolved and particulate samples from the St. Lawrence River from Cornwall, Ontario to L'île d'Orléans, Québec. The authors did not detect 1,4-dichlorobenzene in the particulate phase, however, the compound was detected in the dissolved phase at a concentration of 2.7 ng/L.

Ongley et al. (1988) reported the presence of 1,4-dichlorobenzene in suspended sediments in the North Saskatchewan River in 1983. Concentrations in suspended sediments ranged from non-detectable to 118 ng/g (dry weight) 1,4-dichlorobenzene in two of nine samples (detection limit not specified).

4.3.6   Effluent and Sludge

Sludges from 15 municipal sludges across Canada were analyzed for organic contaminants from 1980 to 1985 (Webber and Lesage, 1989). 1,4-Dichlorobenzene was detected in one third of the samples of sludge from Winnipeg, Manitoba. Maximum and median concentrations of 1 500 ng/g and 300 ng/g (dry weight) 1,4-dichlorobenzene, respectively, were detected (Table 4-5).

A field monitoring program of influent, final effluent and raw and treated sludges from 37 Ontario water pollution control plants was carried out by the Ontario Ministry of the Environment's MISA program in 1987 (OME, 1988). 1,4-Dichlorobenzene was detected in two of 37 municipal water pollution control plants sampled at a maximum concentration of 2 644 ng/g in treated sludge.

4.3.7   Biota

1,4-Dichlorobenzene was detected in amphipods, oligochaetes, clams and fish in studies conducted in the early 1980s. No studies were available on levels of 1,4-dichlorobenzene in wild mammals, reptiles, amphibians or birds in Canada (Table 4-6).

In a study of the effects of sewage effluent on biota at the Clover Point outfall, near Victoria BC, EVS Consultants (1992) reported that 1,4-dichlorobenzene was below the detection limit (5 µg/kg) in mussel (Modiolus sp.) tissue.

Based on studies conducted in the Great Lakes in the early 1980s, the concentrations of 1,4-dichlorobenzene in lake trout (Salvelinus namaycush) and rainbow trout (Oncorhynchus mykiss) ranged between 1 and 4 ng/g (wet weight) (Oliver and Nicol, 1982). Higher concentrations of 1,4-dichlorobenzene were reported in sediment-dwelling organisms collected from five sites in western

Lake Ontario in April and July, 1981 (Fox et al., 1983). Levels of 1,4-dichlorobenzene ranged from not detectable to 370 ng/g (dry weight) in amphipods (Pontoporeia hoyi and Gammarus sp.) and from non detectable to 630 ng/g (dry weight) in oligochaetes (Tubifex tubifex) (detection limits not specified). 1,4-Dichlorobenzene was not detected in mysids and lake trout. Worms (Tubifex tubifex and Limnodrilus hoffmeisteri) exposed to contaminated sediments from Toronto Harbour in Lake Ontario were found to rapidly accumulate 1,4-dichlorobenzene (Oliver, 1987b). Worms were analyzed after 4, 11, 39 and 79 days. A mean concentration of 200 ng/g (dry weight) 1,4-dichlorobenzene was measured in the worms after 4 days and decreased to 60 ng/g after 79 days. 1,4-Dichlorobenzene was found in clams collected in the area of a outfall from a chlor-alkali chemical plant near Pictou, Nova Scotia. MacLaren Marex Ltd. (1979) reported levels of total 1,3- and 1,4-dichlorobenzene ranging from 310 to 590 ng/g (dry weight) in clams.

No data were available on the effects of 1,4-dichlorobenzene on aquatic vascular plants. No studies were available on levels of 1,4-dichlorobenzene in wild mammals or birds.

5.0 POPULATION EXPOSURE

5.1 Exposure of Wildlife Populations

In general, three routes of exposure may be of concern for wildlife: oral, inhalation and dermal. Oral exposures might occur via ingestion of contaminated food (e.g. aquatic prey), or water or incidental ingestion of contaminated media (e.g. soils, sediment). Inhalation of vapours or particulates might be a significant route of exposure for animals active near point sources. Dermal exposures are likely to be most significant for burrowing mammals (i.e., via contact with contaminated soils) and animals that spend considerable amounts of time partially submerged in contaminated surface waters. Wildlife can also be affected indirectly by removal of their food resources if levels in the environment are sufficiently high to be toxic to plants or invertebrates.

6.0 TOXICOKINETICS
(Refer to supporting documentation from Health Canada)

7.0 MAMMALIAN TOXICOLOGY
(Refer to supporting documentation from Health Canada)

8.0 EFFECTS ON HUMANS
(Refer to supporting documentation from Health Canada)

9.0 EFFECTS ON THE ECOSYSTEM

The effects-related information for 1,4-dichlorobenzene includes acute and chronic data from a number of aquatic species. The results of aquatic toxicity tests of 1,4-dichlorobenzene are presented in Tables 9-1 to 9-5. 1,4-Dichlorobenzene is a volatile compound, and therefore, aquatic tests conducted under closed, static or continuous flow conditions were considered reliable. Information on toxicity to terrestrial organisms is very limited. However, the toxicity of 1,4-dichlorobenzene to terrestrial organisms can be assessed by extrapolation from the results of toxicity studies conducted with laboratory mammals.

The mode of toxic action of 1,4-dichlorobenzene is not well understood. The mechanism has been described as a non-specific narcotic or anaesthetic action, controlled by the organism/water partitioning properties of the chemical (Abernethy et al., 1988). It appears that the effect occurs when a target site (thought to be the lipid membrane) accumulates a sufficient volume of chemical to interfere with normal structure and function.

9.1 Aquatic Systems and Components

9.1.1 Bacteria and Primary Producers

Studies on the effects of 1,4-dichlorobenzene on bacteria and

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primary producers are presented in Table 9-1.

The response of _Photobacterium_ _phosphoreum_ to 1,4-dichlorobenzene in a 30 minute Microtox test was measured by Ribo and Kaiser (1983). The reduction of light emitted by _Photobacterium_ _phosphoreum_ when exposed to 1,4-dichlorobenzene was 5.33 mg/L. Inhibition of growth (96-h EC50) and photosynthesis (3-h EC50) at 1.6 mg/L and 5.2 mg/L, respectively, were observed in the alga _Selenastrum_ _capricornutum_ (Calamari et al., 1982; Calamari et al., 1983). The primary productivity of the freshwater green alga _Ankistrodesmus_ _falcatus_ exposed to 1,4-dichlorobenzene was monitored by measuring the uptake of ^{14}C-carbonate by cells over a 4-h period (Wong et al., 1984). A 4-h EC50 value of 19.9 mg/L was reported.

The effect of 1,4-dichlorobenzene on the diatom _Cyclotella_ _meneghiniana_ using percent reduction of DNA was reported by Figueroa and Simmons (1991). The use of biochemical parameters such as nucleic acids measurement is an alternative method of determining the effects of contaminants on cellular components related to growth, biosynthesis and reproduction. Algal DNA reduction occurred after 48 h exposure of _Cyclotella_ _meneghiniana_ to 34.30 mg/L 1,4-dichlorobenzene.

9.1.2   Invertebrates

9.1.2.1   Acute Toxicity

A comparison of aquatic invertebrate sensitivities to 1,4-dichlorobenzene indicated that the freshwater invertebrate _Daphnia_ _magna_ was the most sensitive invertebrate species reported in the literature (Table 9-2). Results of acute toxicity tests using the most acceptable laboratory procedures for the _Daphnia_ ranged from a 48-h immobilization concentration of 0.74 mg/L in a closed, static system to a 48-h LC50 of 2.2 mg/L under closed, static conditions (Canton et al., 1985). In an acute toxicity study with the midge (_Tanytarsus_ _dissimilis_), Call et al. (1983) reported an LC50 value (48-h) of 13.0 mg/L for exposure to 1,4-dichlorobenzene under static, measured conditions.

In studies with saltwater invertebrates, toxicity values were considerably higher, with LC50 values ranging from 60.0 mg/L (96-h LC50) (Curtis et al., 1979) for the grass shrimp (_Palaemonetes_ _pugio_) in static, unmeasured tests to 13.7 mg/L (24-h LC50) for the _Artemia_ nauplii (Abernethy et al., 1986).

9.1.2.2   Chronic Toxicity

Impairment of reproduction was the most sensitive endpoint of toxic stress reported in the literature during chronic toxicity test of 1,4-dichlorobenzene to aquatic invertebrates (Table 9-3).

Under closed, static conditions, Calamari et al. (1982) reported a reduction in fertility in Daphnia magna at 0.40 mg/L (LOEC) 1,4-dichlorobenzene after 28 days while 0.22 mg/L was the no-observed effect concentration (NOEC). In a similarly conducted study performed on Daphnia, Calamari et al. (1983), reported a 14-d EC50 (reduced fertility) of 0.93 mg/L in Daphnia magna. In a 21-day reproduction test on Daphnia, Kuhn et al. (1989) reported a highest no-effect concentration (NOEC) of 0.50 mg/L.

Using calculated quantitative structure activity relationships (QSARs), a number of authors determined chronic responses of Daphnia magna to 1,4-dichlorobenzene (Hermens et al., 1984; Hermens et al., 1985; De Wolf et al., 1985). The highest concentration of 1,4-dichlorobenzene which did not produce a significant effect (16-day NOEC) on reproduction of Daphnia magna was 0.58 mg/L (Hermens et al., 1985). Hermens et al. (1984) determined reproductive effects in Daphnia magna at 0.47 mg/L (16-d EC50) and lethal effects at 1.5 mg/L (16-day LC50). However, it should be noted that chemical analysis showed that for 1,4-dichlorobenzene, between 80 and 110% was present in the test solutions and the NOEC value was not corrected for these measured concentrations. When corrected for the actual dose, the 16-day NOEC (reproduction) is 0.46 and 0.64 mg/L, respectively.

9.1.3   Fish

9.1.3.1   Acute Toxicity

The most sensitive endpoint for acute toxicity to fish was observed in rainbow trout (Oncorhynchus mykiss), at concentrations ranging from 1.18 mg/L (48-h LC50) to 1.48 mg/L (48-h LC50) (Ahmad et al., 1984; Call et al., 1983; U.S. EPA, 1980; Calamari et al., 1983) (Table 9-4). Abnormal swimming and loss of equilibrium for the rainbow trout (Oncorhynchus mykiss) were observed at 1.10 mg/L (96-h EC50) (Ahmad et al., 1984). The American flagfish (Jordanella floridae), fathead minnow (Pimaphales promelas), sheepshead minnow (Cyprinodon variegatus), and zebra fish (Brachydanio rerio), were less sensitive to exposure to 1,4-dichlorobenzene. Under flow-through, unmeasured conditions with one-week and juvenile (2 to 4 month old) American flagfish, ATRG (1987) reported 96-h LC50s of 2.45 and 2.05 mg/L, respectively. For fathead minnows (Pimaphales promelas), 96-h LC50s ranging from 4.0 to 4.2 mg/L were reported by Veith et al. (1983), Carlson and Kosian (1987) and Ahmad et al. (1984). Exposure of fathead minnow embryos (Pimaphales promelas) to 8.7 mg/L of 1,4-dichlorobenzene resulted in premature hatching and deformities after three days (Carlson and Kosian, 1987). Mayes et al. (1983) demonstrated through acute testing (96-h LC50) of three age groups of fathead minnows (Pimephales promelas), that fry (10 to 15 days) were more sensitive than juvenile (30 to 35 days) and adults (60 to 100 days) to 1,4-dichlorobenzene. 96-h LC50 values of 3.6, 14.2 and 11.7 mg/L were reported for tests conducted in closed, static conditions

for fry, juvenile and adult fathead minnows, respectively. A similar LC50 value was observed for the zebrafish (Brachydanio rerio), with a 48-h LC50 of 4.25 mg/L (Calamari et al., 1982).

9.1.3.2 Chronic Toxicity

Growth and survival were the most sensitive endpoints in chronic toxicity tests with fathead minnow (Pimephales promelas) and zebrafish (Brachydanio rerio) (Table 9-5). Effects on embryo-larval stages of fathead minnows under flow-through, measured conditions was studied by Ahmad et al. (1984). The estimated maximum acceptable toxicant concentration (MATC) for fathead minnows exposed to 1,4-dichlorobenzene during a 32-day early-life stage toxicity test based on survival and growth was between 0.56 and 1.04 mg/L (Ahmad et al., 1984). Carlson and Kosian (1987) estimated chronic toxicities for 32 to 33 day embryo through early juvenile development tests of fathead minnow (Pimephales promelas) exposed to 1,4-dichlorobenzene. A highest no observed effect concentration (NOEC) of 0.57 mg/L and a lowest observed effect concentration (LOEC) of 1.0 mg/L was reported for the fathead minnow. Exposure to 8.7 mg/L 1,4-dichlorobenzene caused deformities to embryos. Retardation of growth was the most sensitive parameter in early life stage toxicity experiments on embryolarval (<6 hours old after spawning) stages of zebrafish (Brachydanio rerio) exposed for a period of 28 days (van Leeuwen et al., 1990). A LC50 of 2.7 mg/L was reported for the early life stage test with the zebrafish. The no observed lethal concentration (NOLC: the highest concentration tested without significant effects on survival) of the zebra fish was 2.1 mg/L and the no observed effect concentration for survival, hatching and growth was 0.65 mg/L 1,4-dichlorobenzene.

Exposure of different stages of rainbow trout (Oncorhynchus mykiss) was carried out in a continuous flow-through system for 14 days from fertilized egg to the 2 week old eating alevin stage, at six concentrations ranging from 0.0018 mg/L to 0.122 mg/L (Calamari et al., 1982). Examination of 25 embryos at each concentration indicated that no macroscopic malformations or histological changes were found at the time of hatching. A 14-day LC50 of 0.80 mg/L was determined for the rainbow trout alevins (Calamari et al., 1982).

9.2 Birds and Mammals

No data are available on wild mammals, thus results from laboratory studies were extrapolated to determine effects on wild mammals.

Studies on birds were scarce. Seventeen-day old White Leghorn Chicken embryos injected with 10 and 40 mg/egg 1,4-dichlorobenzene, after 24 hours showed no increase in porphyrin content of the liver of the chick embryos. One-day old chicks dosed orally with 200 and 800 mg/kg demonstrated a significant increase in the porphyrin

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content of liver and bile at the higher dose. This higher dose is approximately equal to the higher dose given to the embryo (40 mg/egg). Thus the chick embryo is not as sensitive as the chick to the porphyrinogenic action of 1,4-dichlorobenzene (Miranda et al., 1984).

The chronic toxicity of 1,4-dichlorobenzene has been examined in rats and mice following inhalation (Loeser and Litchfield, 1983). They reported increases in liver and kidney weights, urinary protein and coproporphyrin in the high dose group of rats administered 0, 450 and 3 000 mg/m^3 5 h/d, 5 d/week, for 76 weeks. There was no evidence of toxicity in mice following exposure to the same concentrations for 57 weeks. The NOELs in rats and mice are therefore 450 and 3 000 mg/m^3, respectively.

In studies on the embryotoxicity and foetotoxicity of 1,4-dichlorobenzene, minor effects were seen, but only at doses which were toxic to the mother (i.e., 2 400 mg/m^3, Hayes et all., 1985; 3000 mg/m^3, Loeser and Litchfield, 1983; 500 mg/kg-bw/d in their diet, Giavini et al., 1986).

Following exposure to 950 mg/m^3 1,4-dichlorobenzene 7 hours/day, 5 days per week, a decrease in body weight gain (>10%) in guinea pigs was observed and in rats, an increase in liver and kidney weights and cloudy swelling and granular degeneration in the liver of rats was observed by Hollingsworth et al. (1956). The NOEL for rats and guinea pigs was ·580 mg/m^3 and 950 mg/m^3 for monkeys, mice and rabbits.

In a chronic inhalation study (dosed for 76 weeks, 5 days/week), the NOELs were 450 mg/m^3 in rats and 3000 mg/m^3 in mice (Loeser and Litchfield, 1983). In a study exposing rats and mice via the oral route for 13 weeks, the NOELs were 300 to 600 mg/kg-bw for male and female rats, respectively, and 338 mg/kg-bw for mice (NTP, 1987).

In studies using oral administration of 1,4-dichlorobenzene, Hollingsworth et al. (1956) reported 2800 mg/kg-bw and 4000 mg/kg-bw were lethal to 100% of guinea pigs and rats, respectively. No deaths occurred at doses of 1600 mg/kg-bw for guinea pigs and 1000 mg/kg-bw for rats.

No empirical data were available regarding adverse effects of 1,4-dichlorobenzene on wildlife due to decreased prey availability or quality. No data are available on the effects of 1,4-dichlorobenzene on terrestrial vascular plants.

9.3   Terrestrial Systems and Components

9.3.1   Terrestrial Toxicity

Toxicity of 1,4-dichlorobenzene to earthworms (Eisenia andrei and Lumbricus rubellus) and soil sorption tests were carried out in two soil types (KOBG - sandy 3.7% organic matter and OECD - peaty sand, 8.1% organic matter) (van Gestel et al., 1991). Greater adsorption of 1,4-dichlorobenzene was observed on the OECD soil than the KOBG soil. The authors reported that the earthworm toxicity of 1,4-dichlorobenzene is predominantly governed by pore water concentrations. Based on the soil pore water phase, fourteen day toxicity values of 17.8 (KOBG) and 51.0 mg/L (OECD) for Eisenia andrei and 26.2 (KOBG) and 228.7 mg/L (OECD) for Lumbricus rubellus were reported.

No data were available on the effects of 1,4-dichlorobenzene on terrestrial vascular plants, and terrestrial invertebrates.

9.4 Sediment Toxicity

No empirical data were available on the toxicity of 1,4-dichlorobenzene to benthic biota in freshwater, estuarine, or marine sediments in Canada.

Marine

EVS Consultants (1992) reported indirect evidence of adverse biological effects to sediment benthos in a study on the effects of municipal effluents of the City of Victoria (B.C.) on the marine environment. The juvenile polychaete Neanthes sp. was significantly affected (growth reduction) at the outfall (Macaulay Point) and at three stations 100 m away and adverse effects to bivalve larvae development (development from a fertilized egg to a fully developed larva) were observed at all stations tested. Although these effects could not be attributed to specific contaminants, EVS Consultants (1992) reported concentrations of 1,710 and 139 µg/kg 1,4-dichlorobenzene at the outfall of the effluent and at stations 100 meters away, respectively.

Barrick et al. (1988) reported effect thresholds for 1,4-dichlorobenzene in marine sediment using the Apparent Effects Threshold (AET) approach. This method involves the comparison of field data on chemical concentrations in sediments and indicators of bioavailability (benthic infaunal analysis, toxicity steps) to estimate the concentration of a chemical above which statistically significant biological effects (relative to reference sediments) are always expected (Barrick et al., 1988). Specifically, this approach was used to estimate the concentration of contaminants in sediment collected in a contaminated area in Puget Sound, Washington State, above which significant (p <0.05) biological effects to marine benthic infauna (Mollusca, Polychaeta, Crustacia) were consistently observed. The marine sediment AET for benthic organisms for 1,4-dichlorobenzene was reported to be 160 ng/g (normalized to 1% total organic carbon). However, as noted by U.S. EPA (1989), the AET approach is not recommended for the development

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of sediment quality criteria to be applied widely due to its site-specific nature, and its inability to describe cause and effect relationships.

Freshwater

Although no empirical freshwater sediment toxicity data were available for 1,4-dichlorobenzene, results from the equilibrium partitioning approach were used to establish effect threshold levels. These values represent concentrations of chemicals in sediment that are expected to be associated with adverse biological effects (Di Toro et al., 1991). This method involves converting toxic threshold concentrations for pelagic organisms to toxic threshold concentrations in sediment which are then compared to sediment concentrations measured in the environment. Water column and benthic species are assumed to be similarly sensitive to 1,4-dichlorobenzene (Di Toro et al., 1991). Additional assumptions built into this approach are that: 1) nonionic organic chemicals such as 1,4-dichlorobenzene are associated with sediment organic carbon, 2) partitioning of chemicals between the organic carbon and sediment pore water is at equilibrium, and 3) ingestion is not a route of exposure.

The effect threshold in sediment (C_s) was calculated based on the equilibrium relationship:

$$C_s, oc = C_s/f_{oc} = K_{oc} \times C_d$$

formulated by Di Toro et al. (1991). C_s, oc is the organic carbon-normalized sediment concentration, and C_d is the dissolved concentration of 1,4-dichlorobenzene that caused effects in the most sensitive organism tested. The organic carbon fraction (f_{oc}) of the sediment was set at 0.035, corresponding to levels measured in the most severely contaminated areas of western Lake Ontario. An estimated effect level of 480 μg/g (the level which could adversely effect sensitive benthic species) was calculated for C_d using a chronic effect level (LOEL) for dissolved 1,4-dichlorobenzene for freshwater organisms of 400 μg/L (28-day LOEL reduced fertility in _Daphnia magna_) (Calamari et al., 1982) and a K_{oc} of 1 030 (Mackay et al., 1992).

van Leeuwen et al. (1992) used quantitative structure-activity relationships and equilibrium partitioning modelling to predict the concentration of 1,4-dichlorobenzene at which 95% of the species in marine or freshwater benthic communities are unlikely to be affected. The authors calculated an effect threshold level of 5336 ng/g (d.w.) for sediments with an organic carbon content of 5%, or 1 067 ng/g for sediments with an organic carbon content of 1%.

Although use of equilibrium partitioning is becoming increasingly common in sediment quality assessment (Di Toro et al.,

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1991), the method has undergone only preliminary field validation (Chapman, 1989; Adams et al., 1992), and thus is of limited use in a regulatory context.

10.0 OTHER PERTINENT DATA

10.1 Greenhouse effects

Like other trace gases that are known to contribute to global warming (Ramanathan et al., 1985), 1,4-dichlorobenzene is volatile at tropospheric temperatures and absorbs infrared radiation in the 7 to 13 μm wavelength region (Ramanathan et al., 1985; Sadtler Research Laboratories, 1982). However, because 1,4-dichlorobenzene is removed from the atmosphere by photooxidation (mean half-life of approximately 3 weeks), and steady state concentrations in the atmosphere are low (mean 0.92 μg/m^3), 1,4-dichlorobenzene is not expected to have a significant direct effect on global warming (Hengeveld, pers. comm., 1992). However, the degradation of 1,4-dichlorobenzene through reactions with the OH radical can contribute to lower concentrations of OH which is an important sink for other greenhouse gases, particularly methane (IPCC, 1990). Therefore, its release into the atmosphere can indirectly contribute to increased global warming. The magnitude of this effect is difficult to estimate because of the complexity of the chemistry, however, it is believed to be small (Hengeveld, pers. commun., 1992).

10.2 Ozone Formation/Depletion

Because of the low concentrations found in the atmosphere in Canada, the contribution of 1,4-dichlorobenzene to the ground-level ozone-forming potential of volatile organic compounds (VOCs) mixtures is insignificant (Dann, pers. comm., 1992).

Substances involved in the depletion of stratospheric ozone are generally halogenated, insoluble in water, and persistent in the atmosphere allowing movement to the stratosphere where they are degraded by high energy, short wavelength ultraviolet radiation (Firor, 1990). 1,4-Dichlorobenzene has a short residence time in the atmosphere due to the combined action of photooxidation and atmospheric washout and is therefore not associated with the depletion of stratospheric ozone.

12. REFERENCES

Abernethy, S.G., A.M. Bobra, W.Y. Shiu, P.G. Wells and D. Mackay. 1986. Acute lethal toxicity of hydrocarbons and chlorinated hydrocarbons to two planktonic crustaceans: the key role of organism-water partitioning. Aquat. Toxicol., 8(3): 163-174.

Abernethy, S.G., D. Mackay, and L.S. McCarty. 1988. "Volume fraction" correlation for narcosis in aquatic organisms: the key role of partitioning. Environ. Toxicol. Chem., 7: 469-481.

Adams, W.J., R.A. Kimerle, J.W. Barnett, Jr. 1992. Sediment quality and aquatic life assessment. Environ. Sci. Technol. 26(10): 1865-1875.

ADS associes ltee. 1990. Usine de Napierville. Caractérisation et mesures de restauration du fossé de drainage. ADS associés ltée, Montréal, Quebec, Projet 17-36-275-001. pp. 25.

Ahmad, N., D. Benoit, L. Brooke, D. Call, A. Carlson, D. DeFoe, J. Huot, A. Moriarty, J. Richter, P. Shubat, G. Veith and C. Wallbridge. 1984. Aquatic toxicity tests to characterize the hazard of volatile organic chemicals in water: a toxicity data summary - parts I and II. U.S. Environmental Protection Agency, Duluth, Minnesota. EPA-600/3-84-009.

(ATRG) Aquatic Toxicity Research Group. 1987. Aquatic toxicity studies of multiple organic compounds. Part I: chlorinated benzenes and chlorinated phenols summary report. (Interim). Ontario Ministry of the Environment and Environment Canada. 277 pp.

Atkinson, R., S.M. Aschmann, A.M. Winer and J.N. Pitts. 1985. Atmospheric gas phase loss processes for chlorobenzene, benzotrifluoride and 4-chlorobenzotrifluoride, and generalization of predictive techniques for atmospheric lifetimes of aromatic compounds. Arch. Environ. Contam. Toxicol., 14: 417-425.

Bachmann, A., W.de Bruin, J.C. Jumelet, H.H.N. Rijnaarts and A.J.B. Zehnder. 1988. Aerobic biomineralization of alpha-hexachlorocyclohexane in contaminated soil. Appl. Environ. Microbiol. 54(2): 548-554.

Barber, L.B. 1988. Dichlorobenzene in ground water: evidence for long-term persistence. Ground Water. 26(6): 696-702.

Barber, L.B., E.M. Thurman, and M.P. Schroeder. 1988. Long-term fate of organic micropollutants in sewage-contaminated groundwater. Environ. Sci. Technol. 22(2): 205-211.

Barker, J.F., J.S. Tessmann, P.E. Plotz and M. Reinhard. 1986. The

organic geochemistry of a sanitary landfill leachate plume.

Barrick, R., S. Becker, L. Brown, H. and R. Pastorok. 1988. Volume I. Sediment quality values refinement: 1988 update and evaluation of Puget Sound AET. PTI Environmental Services. Bellevue, WA. 158 pp.

Barrows, M.E., S.R. Petrocelli and K.J. Macek. 1980. Bioconcentration and elimination by bluegill sunfish (Lepomis macrochirus). In: Dynamics, Exposure and Hazard Assessment of Toxic Chemicals, R. Haque, ed., Ann Arbor Science, Michigan, pp. 379-392.

Bosma, T.N.P., J.R. van der Meer, G. Schraa, M.E. Tros and A.J.B. Zehnder. 1988. Reductive dechlorination of all trichloro- and dichlorobenzene isomers. FEMS Microbiol. Ecol. 53: 223-229.

Bouwer, E.J., P.L. McCarty and J.C. Lance. 1981. Trace organic behaviour in soil columns during rapid infiltration of secondary wastewater. Water Res., 15: 151-159.

Bouwer, E.J., and P.L. McCarty. 1982. Removal of trace chlorinated organic compound. Environ. Sci. Technol. 16(12): 836-843.

Bouwer, E.J. and P.L. McCarty. 1983. Transformations of halogenated organic compounds under denitrification conditions. Appl. Environ. Microbiol. 45(4): 1295-1299.

Bouwer, E.J. and P.L. McCarty. 1984. Modelling of trace organics biotransformation in the subsurface. Ground Water, 22(4): 433-440.

Bouwer, E.J., P.L. McCarty, H. Bouwer and R.C. Rice. 1984. Organic contaminant behaviour during rapid infiltration of secondary wastewater at the Phoenix 23rd Avenue project. Water Res. 18(4): 463-472.

Bouwer, E.J. 1985. Secondary utilization of trace halogenated organic compounds in biofilms. Environ. Prog., 4(1): 43-45.

Bouwer, E.J. and P.L. McCarty. 1985. Utilization rates of trace halogenated organic compounds in acetate-grown biofilms. Biotech. Bioeng., 27: 1564-1571.

Calamari, D. S. Galassi and F. Setti. 1982. Evaluating the hazard of organic substances on aquatic life: the paradichlorobenzene example. Ecotoxic. Environ. Saf. 6: 369-378.

Calamari, D., S. Galassi, F. Setti and M. Vighi. 1983. Toxicity of selected chlorobenzenes to aquatic organisms. Chemosphere, 12(2): 253-262.

Call, D.J. L.T. Brooke, M.L. Knuth, S.H. Poirier and M.D. Hoglund. 1985. Fish subchronic toxicity prediction model for industrial organic chemicals that produce narcosis. Environ. Toxicol. Chem. 4:335-341.

Callahan, M., M. Slimak, N. Gabel, I. May, C. Fowler, R. Freed, P. Jennings, R. Durfee, F. Whitmore, B. Maestri, W. Mabey, B. Holt and C. Gould. 1979. Water-related environmental fate of 129 priority pollutants, vol. II. Office of Water Planning and Standards/Office of Water and Waste Management, United States Environmental Protection Agency, Section V (EPA 440/4-79-029b).

Camford Information Services. 1991. Chlorobenzene CPI product profile. Don Mills, Ontario. 4 pp.

Canton, J.H., W. Sloof, H.J. Kool, J. Struys, TH. J.M. Pouw, R.C.C. Wegman and G.J. Piet. 1985. Toxicity, biodegradability and accumulation of a number of Cl/N-containing compounds for classification and establishing water quality criteria. Reg. Toxicol. Pharmacol., 5: 123-131.

Carlson, A.R. and P.A. Kosian. 1987. Toxicity of chlorinated benzenes to fathead minnows (Pimephales promelas). Arch. Environ. Contam. Toxicol., 16: 129-135.

Chan, C.H. and J. Kohli. 1987. Surveys of trace contamination in the St. Clair River, in 1985. Environment Canada, Science Series No. 158, Inland Waters Directorate, Ontario Region, National Water Quality Branch Laboratory, Canada Centre for Inland Waters, Burlington, Ontario.

Chapman, P.M. 1989. Current approaches to developing sediment quality criteria. Environ. Toxicol. Chem., 8: 589-599.

Chemical Marketing Reporter. 1990. Chemical profile p-dichlorobenzene. July 16, 1990.

Chiou, C.T., P.E.. Porter and D.W. Schmedding. 1983. Partition equilibria of nonionic organic compounds between soil organic matter and water. Environ. Sci. Technol. 17(4): 227-231.

Curtis, M.W., T.L. Copeland and C.H. Ward. 1979. Acute toxicity of 12 industrial chemicals to freshwater and saltwater organisms. Water Res. 13: 137-141.

Curtis, M.W., and C.H. Ward. 1981. Aquatic toxicity of forty industrial chemicals: testing in support of hazardous substance spill prevention regulation. J. Hydrology, 51: 359-367.

Dann, T. 1985. unpubl. Ambient air concentrations of volatile

organic compounds in Toronto and Montreal. Pollution Measurement Division, Technology Development and Technical Services Branch, River Road Environmental Technology Centre, Environment Canada. pp. 83 + appendices.

Dann, T. 1992. Personal communication. Pollution Measurement Division, Conversation and Protection. Environment Canada.

Davis, T. 1991. Personal communication. Pesticides Directorate, Agriculture Canada.

Demirjian, Y.A., A.M. Joshi and T.R. Westman. 1987. Fate of organic compounds in land application of contaminated municipal sludge. J. Water Poll. Control Fed. 59(1): 32-38.

De Bont, J.A.M., M.A.W., Vorage, S. Hartmans, and W.J.J. van den Tweel. 1986. Microbial degradation of 1,3-dichlorobenzene. Appl. Environ. Microbiol., 52(4): 677-680.

De Wolf W., J.H. Canton, J.W. Deneer, R.C.C. Wegman and J.L.M. Hermens. 1988. Quantitative structure-activity relationships and mixture-toxicity studies of alcohols and chlorohydrocarbons: reproducibility of effects on growth and reproduction of Daphnia magna. Aquat. Toxicol. 12(1):39-49.

Di Toro, D.M., C.S. Zarba, D.J. Hansen, W.J. Berry, R.C. Swartz, C.E. Cowan, S.P. Pavlou, H.E. Allen, N.A. Thomas, and P.R. Paquin. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. Environ. Toxicol. Chem., 10: 1541-1583.

Durham, R.W. and B.G. Oliver. 1983. History of Lake Ontario contamination from the Niagara River by sediment radiodating and chlorinated hydrocarbon analysis. J. Great Lakes Res. 9(2): 160-168.

Elder, V.A., B.L. Proctor, and R.A. Hites. 1981. Organic compounds found near dump sites in Niagara Falls, New York. Environ. Sci. Technol., 15(10): 1237-1243.

Ellington, J.J., F.E. Stancil, W.D. Payne, and D.C. Trusty. 1988. Measurement of hydrolysis rate constants for evaluation of hazardous waste land disposal. Volume 3: Data on 70 chemicals. Environmental Research Laboratory, U.S. Environmental Protection Agency, EPA/600/3-88/-28.

Environment Canada. 1985a. The national incinerator testing and evaluation program: two-stage combustion (Prince Edward Island). Summary Report. Urban Activities Division, Environmental Protection Service, Environment Canada. Report EPS/UP/1, pp. 85.

Environment Canada. 1985b. Ambient air concentrations of volatile organic compounds in Toronto and Montreal. River Road Environmental Technology Centre. Ottawa, Ontario, pp. 83.

Environment Canada. 1988. National incinerator testing and evaluation program: environmental characterization of mass burning incinerator technology at Quebec City. Environmental Protection, Environment Canada. Report EPS 3/UP/5. pp. 97.

Environment Canada. 1983. unpubl. Chlorobenzene use patterns. Use Patterns Division, Commercial Chemicals Branch, Environment Canada. pp. 5.

Environment Canada. 1989. Environmental assessment of the Canadian textile industry. Report EPS 5/TX/1. Chemical Industries Division, Industrial Programs Branch, Environment Canada. 104 pp.

Environment Canada. 1990 (unpubl.). Imports of chlorobenzenes. Commercial Chemicals Branch. Environment Canada.

Environment Canada. 1991 (unpubl.). Update and summary report. Measurement program for toxic contaminants in Canadian urban air. River Road Environmental Technology Centre. PMD 91-2.

Environment Canada. 1992. NAQUADAT database for 1,4-dichlorobenzene.

EVS Consultants. 1992. Sediment and related investigations off the Macaulay and Clover Point sewage outfalls. EVS Consultants, North Vancouver, B.C., pp. 193

Fanning, M.L., D.J. Jones, D.W. Larson, and R.G. Hunter. 1989. Environmental monitoring 1987 Iona deep sea outfall project. Beak Associates Consulting, Vancouver, B.C.

Fathepure, B.Z., J.M. Tiedje, and S.A. Boyd. 1988. Reductive dechlorination of hexachlorobenzene to tri-and dichlorobenzenes in anaerobic sewage sludge. Appl.. Environ. Microbiol. 54(2): 327-330.

Figueroa, I.D.C. and M.S. Simmons. 1991. Structure-activity relationships of chlorobenzenes using DNA measurement as a toxicity parameter in algae. Environ. Toxicol. Chem. 10: 323-329.

Firor, J. 1990. The Changing Atmosphere - A Global Challenge. Yale University Press, New Haven & London.

Foratek International Inc. 1987. Étude de caractérisation à l'usine de Recochem Inc. à Napierville. Foratek International Inc., Quebec, Quebec.

Foratek Inc. 1989. Caractérisation complementaire à l'usine de Recochem Inc. à Napierville. Foratek Inc., Quebec, Quebec.

Fox, M.E., J.H. Carey and B.G. Oliver. 1983. Compartmental distribution of organochlorine contaminants in the Niagara River and the western basin of Lake Ontario. J. Great Lakes Res., 9(2): 287-294.

Garrison, A.W., and D.W. Hill. 1972. Organic pollutants from mill persist in downstream waters. American Dyestuff Reporter: 21-25.

Galassi, S. and M. Vighi. 1981. Testing toxicity of volatile substances with algae. Chemosphere, 10(10): 1123-1126.

Germain, A., and C. Langlois. 1988. Contamination des eaux et des sediments en suspension du fleuve Saint-Laurent par les pesticides organochlores, les biphenyls polychlores et d'autres contaminants organiques prioritaires. Water Poll. Res. J. Canada, 23)4): 602-614.

Gersich, F.M., F.A. Blanchard, S.L. Applegath and C.N. Park. 1986. The precision of daphnid (Daphnia magna strauss, 1820) static acute toxicity tests. Arch. Environ. Contam. Toxicol., 15: 741-749.

Giavini, E., Boccia, M.L., Prati, M., and Vismara, C. 1986. Teratologic evaluation of p-dichlorobenzene in the rat. Bull. Environ. Contam. Toxicol. 37: 164-168

Hannah, S.A., B.M. Austern, A.E. Eralp and R.H. Wise. 1986. Comparative removal of toxic pollutants by six wastewater treatment processes. J. Water Poll. Control Fed., 58(1): 27-34.

Haider, K., G. Jagnow, R. Kohnen, and S.U. Lim. 1981. Degradation of chlorinated benzenes, phenols and cyclohexane derivatives by benzene- and phenol-utilizing soil bacteria under aerobic conditions. In: Decomposition of Toxic and Nontoxic Organic Compounds in Soils, M.R. Overcash (ed.). Ann Arbor Science, Ann Arbor Michigan, pp. 207-233.

Hayes, W.C., Hanley, T.R., Gushow, T.S., Johnson, K.A., and John, J.A. 1985. Teratogenic potential of inhaled dichlorobenzenes in rats and rabbits. Fundam. Appl. Toxicol., 5(1): 190-202.

Heitmuller, P.T., T.A. Hollister and P.R. Parrish. 1981. Acute toxicity of 54 industrial chemicals to sheepshead minnows (Cyprinodon variegatus). Bull. Environ. Contam. Toxicol. 27: 596-604.

Hengeveld, H. 1992. Personal communication. Canadian Climate

Centre, Atmospheric Environment Service, Environment Canada.

Hermens, J., H. Canton, P. Janssen and R. de Jong. 1984. Quantitative structure-activity relationships and toxicity studies of mixtures of chemicals with an anaesthetic potency: acute lethal and sublethal toxicity to Daphnia magna. Aquat. Toxicol., 5: 143-154.

Hermens, J., E. Broekhuyzen, H. Canton and R. Wegman. 1985. Quantitative structure-activity relationships and mixture toxicity studies of alcohols and chlorohydrocarbons: effects on growth of Daphnia magna. Aquat. Toxicol. 6:209-217.

Holliger, C., G. Schraa, A.J.M. Stams and A.J.B. Zehnder. Enrichment and properties of an anaerobic mixed culture reductively dechlorinating 1,2,3-trichlorobenzene to 1,3-dichlorobenzene. Appl. Environ. Microbiol. 58(5): 1636-1644.

Hollingsworth, R.L., V.K. Rowe, F. Oyen, H.R. Hoyle, and H. Spencer. 1956. Toxicity of paradichlorobenzene; determinants on experimental animals and human subjects. Arch. Ind. Health, 14: 138-147.

Howard, P.H. 1989. Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Volume I. Large Production and Priority Pollutants. Lewis Publishers, Chelsea, Michigan.

Howard, P.H., G.W. Sage, W.F. Jarvis and D.A. Gray. 1990. Handbook of environmental fate and exposure data for organic chemicals. Volume II. Solvents. Lewis Publishers.

Hutchins, S.R., M.B. Tomson and C.H. Ward. 1983. Trace organic contamination of ground water from a rapid infiltration site: a laboratory-field coordinated study. Environ. Toxicol. Chem. 2: 195-216.

Hutchins, S.R., M.B. Tomson, J.T. Wilson and C.H. Ward. 1984. Fate of trace organics during rapid infiltration of primary wastewater at Fort Devens, Massachusetts. Water Res. 18(8): 1025-1036.

(IPCC) Intergovernmental Panel on Climate Change. 1990. Climate change: the IPCC scientific assessment. (Eds.) Houghton, J.T., G. Jenkins and H.H. Ephraums. Cambridge University Press. New York.

Kao, C. and Poffenberger, N. 1979. Chlorinated benzenes. In: Kirk Othmer. Encyclopedia of Chemical Technology. 3rd Edition, Volume 5. John Wiley & Sons, Toronto. pp. 797-808.

King, L. and G. Sherbin. 1986. Point sources of toxic organics to the upper St. Clair River. Water Pollut. Res. J. Canada 21(3):

433-443.

Kirk, P.W.W., H.R. Rogers and J.N. Lester. 1989. The fate of chlorobenzenes and permethrins during anaerobic sewage sludge digestion. Chemosphere 18(9/10): 1771-1784.

Konemann, H. 1981. Quantitative structure-activity relationships in fish toxicity studies. Toxicol. 19: 209-221.

Kuhn, E.P., P.J. Colberg, J.L. Schnoor, O. Wanner, A.J.B. Zehnder and R.P. Schwarzenbach. 1985. Microbial transformations of substituted benzenes during infiltration of river to groundwater: laboratory column studies. Environ. Sci. Technol., 19: 961-968.

Kuhn, R., M. Pattard, K.D. Pernak, and A. Winter. 1989. Results of the harmful effects of water pollutants to Daphnia magna in the 21 day reproduction test. Wat. Res., 23(4): 501-510.

Kuhn, R and M. Pattard. 1990. Results of the harmful effects of water pollutants to green algae (Scenedesmus subspicatus) in the cell multiplication inhibition test. Water Res., 24(1): 31-38.

Kuntz, K.W., and N.D. Warry. 1983. Chlorinated organic contaminants in water and suspended sediments of the lower Niagara River. J. Great Lakes Res., 9(2): 241-248.

LeBlanc, G.A. 1980. Acute toxicity of priority pollutants to water flea (Daphnia magna). Bull. Environ. Contam. Toxicol., 24: 684-691.

Ligocki, M.P., C. Leuenberger and J.F. Pankow. 1985. Trace organic compounds in rain - II. Gas scavenging of neutral organic compounds. Atmospheric Environment, 19(10): 1609-1617.

Loeser, E. and M.H. Litchfield. 1983. Review of recent toxicology studies on p-dichlorobenzene. Fd. Chem. Toxicol., 21(6): 825-832.

Mackay, D. and A.T.K. Yuen. 1983. Mass transfer coefficients for volatilization of organic solutes from water. Environ. Sci. Technol., 17: 211-217.

Mackay, D., W.Y. Shiu and K.C. Ma. 1992. The Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals. Vol. I, Monoaromatic Hydrocarbons, Chlorobenzenes and PCBs. Lewis Publishers, Inc., MI, USA. pp. 697.

MacLaren Marex Inc. 1979. Report on an environmental survey for chlorobenzenes at four coastal sites in Nova Scotia, 18 pp.

Report prepared for Environmental Protection Service, Halifax, MNS. pp. 33.

Maguire, J., and R.J. Tkacz. 1989. Potential underestimation of chlorinated hydrocarbon concentrations in fresh water. Chemosphere, 19(8/9): 1277-1287.

Mansour, M., E. Feicht, and P. Meallier. 1989. Improvement of the photostability of selected substances in aqueous medium. Toxicol. Environ. Chem. 20/21: 139-147.

Marcos, R. 1992 (unpubl.). U.S. EPA TRI release inventory 1989 for PSL substances. Industrial Programs Branch, Environment Canada.

Marsalek, J., and H. Schroeter. 1988. Annual loadings of toxic contaminants in urban runoff from the Canadian Great Lakes basin. Water Poll. Res. J. Canada. 23(3): 360-378.

Martel, R. 1988. Groundwater contamination by organic compounds in Ville Mercier: new developments. In: NATO/CCMS Second International Conference. Demonstration of Remedial Action Technologies for Contaminated Land and Groundwater, Bilthoven, The Netherlands, 7-11 November, pp. 144-177.

Martel, R., and P. Ayotte. 1989. Etat de la situation sur la contamination de la nappe souterraine dans la région de la ville de Mercier. Ministére de l'Environnement du Québec.

Mayes, M.A., H.C. Alexander and D.C. Dill. 1983. A study to assess the influence of age on the response of fathead minnows in static acute toxicity tests. Bull. Environ. Contam. Toxicol., 31: 139-147.

Mayes, M.A., T.J. Shafer and M.G. Barron. 1988. Critical evaluation of the fathead minnow 7-day static renewal test. Chemosphere, 17(11): 2243-2252.

McCall, J.P., D.A. Laskowski, R.L. Swann, and H.J. Dishburger. 1981. Measurement of sorption coefficients of organic chemicals and their use in environmental fate analysis. In: Test Protocols for Environmental Fate and Movement of Toxicants. Proceedings of a Symposium. Association of Official Analytical Chemists. 94th Annual Meeting, October 21-22, 1980.

McCarty, P.L. and M. Reinhard. 1980. Trace organics removal by advanced wastewater treatment. J. Water Poll. Control Fed., 52(7): 1907-1922.

Melcer, H. H. Monteith and S.G. Nutt. 1988. Variability of toxic trace contaminants in municipal sewage treatment plants. Wat. Sci. Tech., 20(4/5: 275-284.

Melcer, H. 1990. Volatile organic emission from municipal sewage treatment plants. Wastewater Technology Centre Newsletter, No. 15. 3 pp.

Merriman, J.C. 1987. Trace organic contaminants in sediment of the international section of the St. Lawrence River, 1981. Technical Bulletin No. 148. Environment Canada, Water Quality Branch, Inland Waters/Lands Directorate, Burlington, Ontario.

Miranda, D.L., J.L. Wang, M.C. Henderson, H.S. Nakaue and D.R. Buhler. 1984. Effects of chlorobenzenes on hepatic porphyrin and drug metabolism in chick embryo and day-old chick. Research Communications in Chemical Pathology and Pharmacology 46: 13-24.

Morrison, R.T. and R.N. Boyd. 1987. Organic chemistry. Fifth edition. New York University, Allyn and Bacon, Inc. Toronto.

NRDIG (Niagara River Data Interpretation Group). 1988. Joint evaluation of upstream/downstream Niagara River monitoring data. 1986-1987. 37 pp.

NRDIG (Niagara River Data Interpretation Group). 1990. Joint evaluation of upstream/downstream Niagara River monitoring data. 1988-1989. 72 pp.

(NTP) National Toxicology Program. 1987. NTP technical report on the toxicology and carcinogenicity studies of 1,4-dichlorobenzene (CAS No. 106-46-7) in F344/N rats and B6C3F1 mice (gavage studies). NTP TR 319, U.S. Dept. of Health and Human Services, Research Triangle Park, North Carolina, pp.198.

OECD. 1979. Chemicals Testing Programme, Expert Group Physical Chemistry. Final Report (Draft), vol. II. Berlin. (Quoted in Rippen, et al., 1984).

Oliver, B.G., E.G. Cosgrove, and J.H. Carey. 1979. Effect of suspended sediments on the photolysis of organics in water. Environ. Sci. Technol. 13(9): 1075-1077.

Oliver, B.G. and K.D. Bothen. 1982. Extraction and clean-up procedures for measuring chlorobenzenes in sediments and fish by capillary gas chromatography. Intern. J. Environ. Anal. Chem. 12: 131-139.

Oliver, B.G. and K.D. Nicol. 1982. Chlorobenzenes in sediments, water, and selected fish from Lakes Superior, Huron, Erie, and Ontario. Environ. Sci. Technol. 16(8): 532-536.

Oliver, B.G. and A.J. Niimi. 1983. Bioconcentration of chlorobenzenes from water by rainbow trout: correlations with

partition coefficients and environmental residues. Environ. Sci. Technol. 17: 287-291.

Oliver, B.G. and M.N. Charlton. 1984. Chlorinated organic contaminants on settling particulates in the Niagara River vicinity of Lake Ontario. Environ. Sci. Technol. 18(12): 903-908.

Oliver B.G. and K.D. Nicol. 1984. Chlorinated contaminants in the Niagara River, 1981-1983. Sci. Total Environ. 39: 57-70.

Oliver, B.G. 1985. Desorption of chlorinated hydrocarbons from spiked and anthropogenically contaminated sediments. Chemosphere. 14(8): 1087-1106.

Oliver, B.G. and C.W. Pugsley. 1986. Chlorinated contaminants in St. Clair River sediments. Water Poll. Res. J. Canada, 21(3): 368-379.

Oliver, B.G., and J.H. Carey. 1986. Photodegradation of wastes and pollutants in aquatic environment. In: Homogeneous and Heterogeneous Photocatalysis, E. Pelizzetti and N. Serpone (eds.). D. Reidel Publishing Company.

Oliver, B.G. 1987a. Fate of some chlorobenzenes from the Niagara River in Lake Ontario. In: Hites, R.A. and S.J. Eisenreich, (ed.), Sources and Fates of Aquatic Pollutants, Advances in Chemistry Series, American Chemical Society, Washington, D.C.

Oliver, B.G. 1987b. Biouptake of chlorinated hydrocarbons from laboratory-spiked and field sediments by oligochaete worms. Environ. Sci. Technol. 21(8): 785-790.

Oliver, B.G. 1992. Personal communication. Zenon Laboratories. Burnaby, British Columbia.

Ongley, E.D., D.A. Birkholz, J.H. Carey and M.R. Samoiloff. 1988. Is water a relevant sampling medium for toxic chemicals? An alternative environmental sensing strategy. J. Environ. Qual., 17(3): 391-401.

(OME) Ontario Ministry of the Environment. 1988. Thirty-seven municipal water pollution control plants. Volume 1. Municipal Industrial Strategy for Abatement (MISA). Interim Report. pp. 97.

(OME) Ontario Ministry of the Environment. 1991a. Preliminary report on the first six months of process effluent monitoring in the MISA pulp and paper sector (January 1,, 1990 to June 30, 1990). Municipal/Industrial Strategy for Abatement (MISA). Water Resources Branch, Ontario Ministry of the Environment. 176 pp.

(OME) Ontario Ministry of the Environment. 1991b. Preliminary report on the second six months of process effluent monitoring in the MISA pulp and paper sector (July 1, 1990 to December 31, 1990). Municipal/Industrial Strategy for Abatement (MISA). Water Resources Branch, Ontario Ministry of the Environment. 159 pp.

(OME) Ontario Ministry of the Environment. 1992 (unpubl.). Twelve month report. Organic manufacturing sector. Municipal Strategy for Abatement (MISA). Municipal/Industrial Strategy for Abatement (MISA). Water Resources Branch, Ontario Ministry of the Environment. 81 pp.

(OME) Ontario Ministry of the Environment. 1992a. Six month monitoring data report organic chemical manufacturing sector (October 1, 1989 to March 31, 1990). Municipal/Industrial Strategy for Abatement (MISA). Water Resources Branch, Ontario Ministry of the Environment. 123 pp.

(OME) Ontario Ministry of the Environment. 1992b. Twelve month monitoring data report. Inorganic chemical sector. Periods covered: December 1, 1989 to November 30, 1990; February 1, 1990 to January 31, 1991. Municipal/Industrial Strategy for Abatement (MISA). MISA Industrial Section, Water Resources Branch, Ontario Ministry of the Environment. 81 pp.

Pankow, J.F., L.M. Isabelle, J.P. Hewetson and J.A. Cherry. 1984a. A syringe and cartridge method for down-hole sampling for trace organics in ground water. Ground Water 22(3): 330-339.

Pankow, J.F., L.M. Isabelle and W.E. Asher. 1984b. Trace organic compounds in rain. 1. Sampler design and analysis by adsorption/thermal desorption (ATD). Environ. Sci. Technol. 18(5): 310-318.

Pankow, J.F., L.M. Isabelle, J.P. Hewetson and J.A. Cherry. 1985. A tube and cartridge method for down-hole sampling for trace organic compounds in ground water. Ground Water. 23(6): 775-782.

Pereira, W.E., C.E. Rostad, C.T. Chiou, T.I. Brinton,, L.B. Barber, D.K. Demcheck and C.R. Demas. 1988. Contamination of estuarine water, biota, and sediment by halogenated organic compounds: a field study. Environ. Sci. Technol., 22(7): 772-778.

Process Industries Canada 1989, 1990 and 1991. Chemical suppliers.

Ramanathan, V., R.J. Cicerone, H.B. Singh, and J.T. Kiehl. 1985. Trace gas trends and their potential role in climate change. J. Geophys. Res. 90(D3): 5547-5566.

Reinhard, M., Goodman, N.L. and Barker, J.F. 1984. Occurrence and

distribution of organic chemicals in two landfill leachate plumes. Environ. Sci. Technol. 18(12): 953-961.

Ribo, J.M. and K.L.E. Kaiser. 1983. Effects of selected chemicals to photoluminescent bacteria and their correlations with acute and sublethal effects on other organisms. Chemosphere 12(11/12): 1421-1442.

Rippen, G., W. Klopffer, R. Frische, and K-O. Gunther. 1984. The environmental model segment approach estimating potential environmental concentrations. Ecotoxicol. Environ. Saf., 8: 363-377.

Sadtler Research Laboratories. 1982. 1,4-Dichlorobenzene. Sadtler Research Laboratories, Division of Bio-Rad Laboratories Inc.

Schraa, G., M. L. Boone, M.S.M. Jetten, A.R.W. van Neerven, P.J. Colberg and A. J.B. Zehnder. 1986. Degradation of 1,4-dichlorobenzene by Alcaligenes sp. strain A175. Appl. Environ. Microbiol., 52(6): 1374-1381.

Schraa, G and J.R. van der Meer. 1987. Biotransformations of di-and trichlorinated benzenes. In: Moody, G.W.and Baker P.B. (eds.). International Conference on Bioreactors and Biotransformations. Gleneagles, Scotland, UK: 9-12 November 1987. Elsevier Appl. Sci. London, UK.

Schwarzenbach, R.P., E.Molner-Kubica., W. Giger and S.G. Wakeham. 1979. Distribution, residence time, and fluxes of tetrachloroethylene and 1,4-dichlorobenzene in Lake Zurich, Switzerland. Environ. Sci. Technol. 13(11): 1367-1373.

Shane, B.S., C.B. Henry, J.H. Hotchkiss, K.A. Klausner, W.H. Gutenmann and D.J. Lisk. 1990. Organic toxicants and mutagens in ashes from eighteen municipal refuse incinerators. Arch. Environ. Contam. Toxicol. 19: 665-673.

Singh, H.B., L.J. Salas, A.J. Smith and H. Shigeishi. 1981. Measurements of some potentially hazardous organic chemicals in urban environments. Atmos. Environ. 15: 601-612.

Singh, H.B., L.J. Salas, R. Stiles, and H. Shigeishi. 1983. Measurements of hazardous organic chemicals in the ambient atmosphere. Environmental Sciences Research Laboratory-RTP, Office of Research and Development, United States Environmental Protection Agency. PB83-156935. EPA/600/09

Sloof, W., 1979. Detection limits of a biological monitoring system based on fish respiration. Bull. Environ. Contam. Toxicol., 23: 517-523.

Spain, J.C. and S.F. Nishino. (1987). Degradation of 1,4-

dichlorobenzene by a _Pseudomonas_ _sp_. Appl. Environ. Microbiolog. 53: 1010-1019.

Spicer, C.W., R. M. Riggin, M.W. Holdren, F.L. DeRoos, and R.N. Lee. 1985. Atmospheric reaction products from hazardous air pollutant degradation. Atmospheric Sciences Research Laboratory, U.S. Environmental Protection Agency, Research Triangle Park. PB85-185841. EPA/600/3-85/028.

Stevens, R.J.J. and M.A. Neilson. 1989. Inter- and intralake distribution of trace contaminants in surface waters of the Great Lakes. J. Great Lakes Res. 15(3): 377-393.

Sylvestre, A. 1987. Organochlorines and polyaromatic hydrocarbons in the St. Lawrence River at Wolfe Island, 1982/84. Technical Bulletin No. 144, Inland Waters/Lands Directorate, Water Quality Branch, Environment Canada. pp. 11.

Tabak, H.H., S.A. Quave, C.I. Mashni and E.F. Barth. 1981. Biodegradability studies with organic priority pollutant compounds. J. Water Poll. Control Fed. 53(10): 1503-1518.

Topping, B. 1987. The biodegradability of para-dichlorobenzene and its behaviour in model activated sludge plants. Wat. Res. 21(3): 295-300.

U.S. EPA. 1978. In-depth studies on health and environmental impacts of water pollutants. U.S. Env. Prot. Agency, Contract No. 68-01-4646. (Also cited in U.S. EPA, 1980a and b).

U.S. EPA. 1980. Ambient water quality criteria for dichlorobenzenes. Office of Water, Regulations and Standards, Criteria and Standards Division, EPA 440/5-80-039 (PB-81-117525). (Secondary Reference)

U.S. EPA. 1982. Human Exposure to Atmospheric Concentrations of Selected Chemicals, Volume II: A Summary of Data on Chlorobenzenes. Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina. PB83-265249. pp.7-4 - 7-52.

U.S. EPA. 1983. Toxicity and metabolism studies with EPA priority pollutants and related chemicals in freshwater organisms. Environmental Research Laboratory-Duluth, United States Environmental Agency, (PB83-263665). 134 pp.

U.S. EPA. 1986. Locating and estimating air emissions from sources of chlorobenzenes. Office of Air Quality Planning and Standards, United States Environmental Agency, 146 pp. (EPA-450/4-84-007m). 135pp. (PB87-189841/REB)

U.S. EPA. 1987. Health advisories for 25 organics. Office of

Drinking Water, United States Environmental Protection Agency, 399 pp. (PB87-235578).

U.S. EPA. 1991. Minimization and control of hazardous combustion byproducts. U.S. Environmental Protection Agency, Research and Development, Risk Reduction Engineering Laboratory, Cincinnati, Ohio, EPA/600/S2-90/039.

van der Meer, J.R., W. Roelofsen, G. Schraa and A.J.B. Zehnder. 1987. Degradation of low concentrations of dichlorobenzenes and 1,2,4-trichlorobenzene by Pseudomonas sp. strain P51 in nonsterile soil columns. FEMS Microbiol. Ecology, 45: 333-341.

van Gestel, C.A.M., W. Ma and C.E. Smit. 1991. Development of QSARs in terrestrial ecotoxicology: earthworm toxicity and soil sorption of chlorophenols, chlorobenzenes and dichloroaniline. The Sci. Total Environ., 109/110: 589-604.

van Leeuwen, C.J., D.M.M. Adema and J. Hermens. 1990. Quantitative structure-activity relationships for fish early life stage toxicity. Aquat. Toxicol., 16: 321-334.

van Leeuwen, C., P.T.J. van der Zandt, T. Aldenberg, H.J.M. Verhaar and J.L.M. Hermens. 1992. Application of QSARs, extrapolation and equilibrium partitioning in aquatic effects assessment. I Narcotic industrial pollutants. Environ. Toxicol. Chem., 11: 267-282.

Veith G.D., K.J. Macek, S.R. Petrocelli and J. Carroll. 1980. An evaluation of using partition coefficients and water solubility to estimate bioconcentration factors for organic chemicals in fish. Aquatic Toxicology, ASTM STP 707. J.G. Eaton, P.R. Parrish and A.C. Hendricks, Eds., American Society for Testing and Materials. pp. 116-129.

Veith, G.D., D.J. Call and L.T. Brooke. 1983. Structure-toxicity relationships for the fathead minnow, Pimephales promelas: narcotic industrial chemicals. Can. J. Fish. Aquat. Sci. 40: 743-748.

Wakeham, S.G., A.C. Davis and J.L. Karas. 1983. Mesocosm experiments to determine the fate and persistence of volatile organic compounds in coastal seawater. Environ. Sci. Technol. 17: 611-617.

Webber, M.D., and S. Lesage. 1989. Organic contaminants in Canadian municipal sludges. Waste Management & Research 7: 63-82.

Wong, P.T.S., Y.K. Chau, J.S. Rhamey and M. Docker. 1984. Relationship between water solubility of chlorobenzenes and their effects on a freshwater alga. Chemosphere, 13(9): 991-996.

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Wilson, J.T., C.G. Enfield, W.J. Dunlap, R.L. Cosby, D.A. Foster, and L.B. Baskin. 1981. Transport and fate of selected organic pollutants in a sandy soil. J. Environ. Qual., 10(4): 501-506.

Zoeteman, B.C.J., K. Harmsen, J.B.H.J. Linders, C.F.H. Morra and W. Sloof. Persistent organic pollutants in river water and grounds water of the Netherlands. Chemosphere, 9: 231-249.

TABLES

Table 2-1.   Physical and Chemical Properties of
                 1,4-dichlorobenzene

PROPERTY		REFERENCE
Molecular Weight (g/mol)	147.01	Mackay et al. (1992)
Melting Point (°)	53.7	Mackay et al. (1992)
Boiling Point (°)	174.6	Mackay et al. (1992)
Density		
(g/ml @ 20°C)	1.2475	Mackay et al. (1992)
Solubility in Water		
(mg/L @ 25°C)	79	Mackay et al. (1992)
Vapour Pressure (Pa)	90.2	Mackay et al. (1992)
Octanol-water Partition		
Coefficient (Log Kow)	3.40 ± 0.1	Mackay et al. (1992)
Henry's Law		
(Pa m^3/mol)	160	Mackay et al. (1992)
Soil Organic		
Distribution		
Coefficient (Log Kom)	2.20	Chiou et al. (1983)

Table 3-1 Use pattern of dichlorobenzenes in Canada (tonnes)[1]

Year	1984	1985	1986	1987	1988	1989	1990	1995f[2]
Domestic production	4000	4500	3600	2500	2100	1900	1900	2500
Imports	1712	1102	1922	3334	3919	3540	3900	3500
Total supply	5712	5602	5522	5834	6019	5440	5800	6000

[1] Camford Information Services, 1991; isomer not specified
[2] forecasted

Table 3-2 Imports of 1,4-dichlorobenzene into Canada[1]

Year	Amount (tonnes)
1990	3,386.2
1989	2,975.0
1988	2,813.1
1987	3,269.2
1986	1,650.5
1985	990.1
1984	1,633.0
1983	2,059.0
1982	2,457.0
1981	3,629.0
1980	4,607.0

[1] Data from Statistics Canada

Table 3-3 Levels of 1,4-Dichlorobenzene in Industrial and Municipal Wastewater in Canada

Location	Year	Concentration (ng/L) Mean	Range min max	Reference
Water Pollution Control Plants				
WPCP ON	1980	660	(484-920)	Oliver and Nicol (1982)
WPCP ON	1986		(ND-2744)	OME (1988)
Galt, WPCP[1] ON		1400		Melcer et al. (1988)
Waterloo WPCP ON		900		Melcer et al. (1988)
Welland WPCP ON		900		Melcer et al. (1988)
Vancouver, BC				
(composite sample)	1987	<240	(<10-390)	Fanning et al. (1989)
(grab sample)	1987		(170-700)	Fanning et al. (1989)
Pulp and Paper Mills				
Fort Francis, ON	1986	ND[2]		Merriman (1988)
Iroquois Falls, ON	1990	40	(<DL[3]-220)	OME (1991a)
Beaver Wood, ON	1990	280	(<DL-1710)	OME (1991a)
James River, ON	1990	1600	(<DL-9600)	OME (1991a)
Malette, ON	1990	90	(<DL-440)	OME (1991a)
Trent Valley, ON	1990	150	(<DL-310)	OME (1991a)
Kenora, ON	1990	50	(<DL-300)	OME (1991b)
Textile Plants				
Cambridge, ON	1985/86	1000		Environment Canada (1989)
Bramalea, ON	1985/86	36000		Environment Canada (1989)
Woodstock, ON	1985/86	71100		Environment Canada (1989)
Magog/Beauharnois, QU	1985/86		(ND-3400)	Environment Canada (1989)

[1] WPCP - Water Pollution Control Plant
[2] DL - <0.1 ng/L
[3] DL - <1.7 µg/L

Table 3-3 cont'd   Levels of 1,4-Dichlorobenzene in Industrial and Municipal Wastewater in Canada

Media/Location	Year	Mean	(Range ng/L) min	max	Reference
Textile Plants					
Truro, NS (1,3-&1,4-DCB)	1979		(7500-9400)		Maclaren Marex Inc. (1979)
Organic Chemical Plant					
Sarnia, ON					
landfill leachate	1985	260			King & Sherbin (1986)
Outfall	1985	13000			King & Sherbin (1986)
Township ditch	1985	5000			King & Sherbin (1986)
loading	1985	0.035 kg/d			King & Sherbin (1986)
Sarnia, ON	1989/90	3500			OME (1992, unpubl.)
loading	1989/90	0.002 kg/d			OME (1992, unpubl.)
Sarnia, ON	1989/90	3900			OME (1992, unpubl.)
loading	1989/90	0.118 kg/d			OME (1992, unpubl.)
Sarnia, ON	1989/90	300			OME (1992, unpubl.)
loading	1989/90	0.052 kg/d			OME (1992, unpubl.)
Sarnia, ON	1989/90			200	OME (1992, unpubl.)
				200	OME (1992, unpubl.)
				200	OME (1992, unpubl.)
				200	OME (1992, unpubl.)
				200	OME (1992, unpubl.)
				500	OME (1992, unpubl.)
				2400	OME (1992, unpubl.)
loading	1989/90	0.365 kg/d			OME (1992, unpubl.)
Maitland, ON	1989/90			300	OME (1992a)
loading	1989/90	0.055 kg/d			OME (1992a)

Table 3-3 cont'd   **Levels of 1,4-Dichlorobenzene in Industrial and Municipal Wastewater in Canada**

Media/Location	Year	Mean	(Range ng/L) min max	Reference
Inorganic Plant				
Niagara Falls, ON	1990/91		1100	OME (1992b)
loading	1990/91	0.004 kg/d		
Niagara Falls, ON	1990/91		1100	OME (1992b)
loading	1990/91	0.003 kg/d		
Niagara Falls, ON	1990/91		1450	OME (1992b)
loading	1990/91	0.005 kg/d		
Sault St. Marie, ON	1990/91		1100	OME (1992b)
loading	1990/91	0.001 kg/d		OME (1992b)

Table 3-4.   Emission factors for 1,4-dichlorobenzene

Emission category	kg 1,4-DCB emitted per tonne 1,4-DCB produced
Process	5.81
Storage	0.41
Fugitive	1.02
TOTAL	7.24

[1]   The above emission factors are only general estimates (U.S. EPA, 1986).

Table 3-5 Total release (tonnes/yr) of 1,4-DCB into air, land and water in the United States reported in the TRI[1] for 1989.

compound	air	water	land	underground injection	offsite	POTW [2]	Fugitive	Total
1,4-DCB	514	3	0.11	0.11	47	12	25	602

[1] Toxic Release Inventory (Marcos, 1992, unpubl.)
[2] POTW - Publicly Owned Treatment Works

Table 4-1   Levels of 1,4-Dichlorobenzene in Air in Canada and the United States

Media/Location	Year	Mean	Range (μg/m³) min max	Reference
Halifax, NS	1988/90	0.6	(0.2-1.9)	Environment Canada (1991, unpubl.)
Montreal, QC				
Pte aux Trembles	1988/90	0.3	(ND-1.5)	Environment Canada (1991, unpubl.)
Ontario St.	1988/90	1.1	(0.3-6.4)	Environment Canada (1991, unpubl.)
Ottawa, ON	1988/90	1.1	(ND-4.7)	Environment Canada (1991, unpubl.)
Toronto, ON				
Junction Triangle	1989/90	0.5	(0.1-2.9)	Environment Canada (1991, unpubl.)
Perth/Ruskin	1989/90	2.9	(0.4-15.7)	Environment Canada (1991, unpubl.)
Stouffville	1989/90	0.47	(ND-2.1)	Environment Canada (1991, unpubl.)
Hamilton, ON	1989/90	0.48	(0.1-2.1)	Environment Canada (1991, unpubl.)
Sarnia, ON	1989/90	0.3	(ND-2.0)	Environment Canada (1991, unpubl.)
Windsor, ON				
University Ave	1989/90	2.0	(ND-14.6)	Environment Canada (1991, unpubl.)
Windsor, ON				
College/Prince	1989/90	0.78	(0.2-2.9)	Environment Canada (1991, unpubl.)
Walpole Island	1989/90	1.3	(ND-3.5)	Environment Canada (1991, unpubl.)
Winnipeg, MAN	1989/90	0.48	(0.09-2.4)	Environment Canada (1991, unpubl.)
Edmonton, AB	1989/90	0.45	(0.07-2.8)	Environment Canada (1991, unpubl.)
Calgary, AB	1989/90	0.5	(0.08-1.7)	Environment Canada (1991, unpubl.)
Vancouver, BC				
Robson Square	1989/90	0.76	(0.2-1.7)	Environment Canada (1991, unpubl.)
Kensington Park	1989/90	0.59	(0.14-1.2)	Environment Canada (1991, unpubl.)
Rocky Point Park	1989/90	0.5	(ND-4.8)	Environment Canada (1991, unpubl.)
Surrey East	1989/90	0.46	(0.17-1.5)	Environment Canada (1991, unpubl.)
Richmond South	1989/90	0.59	(0.1-2.3)	Environment Canada (1991, unpubl.)
Burmount	1989/90	0.51	(ND-2.0)	Environment Canada (1991, unpubl.)
Mahon Park	1989/90	0.79	(0.1-2.6)	Environment Canada (1991, unpubl.)
Burnaby	1989/90	0.22	(0.1-0.4)	Environment Canada (1991, unpubl.)
Montreal, QC				
Pte aux Trembles	1984/85	0.3	(0.1-0.8)[1]	Dann (1985 unpubl.)
Toronto, ON				
Junction Triangle	1984/85	0.37	(ND-2.2)	Dann (1985 unpubl.)

[1] DL - <0.001 μg/m³

Table 4-1 cont'd   Levels of 1,4-Dichlorobenzene in Air in Canada and the United States

Media/Location	Year	Mean (Range) min max	Reference
Air (μg/m^3)			
Atmospheric gas (ng/m^3)			
Portland, OR	1984	120 73-170	Ligocki et al. (1985)
Dissolved rain (ng/L)			
Portland, OR	1984	4.8 3.3-7.0	Ligocki et al. (1985)
Portland, OR (spring)	1982	0.66 0.45-1.2	Pankow et al. (1984b)
Portland, OR (winter)	1982	5.5 3.8-9.2	Pankow et al. (1984b)

Table 4-2 Levels of 1,4-Dichlorobenzene in Canadian Surface Water

Media/Location	Year	Mean	(Range) min max	Reference

Surface Water (ng/L)

Media/Location	Year	Mean	(Range) min max	Reference
Great Lakes Basin urban runoff	1979/83	89		Marsalek & Schroeter (1988)
Rainy R., ON	1986	ND		Merriman (1988)
Lake Superior	1986	0.34	(0.220-0.530)	Stevens & Neilson (1989)
Lake Huron	1980	4	(3-6)	Oliver & Nicol (1982)
	1986	0.29	(0.198-0.396)	Stevens & Neilson (1989)
Georgian Bay	1986	0.38	(0.236-0.631)	Stevens & Neilson (1989)
St. Clair R.	1985		(ND-1.73)	Chan & Kohli (1987)
Lake Erie	1986	1.14	(0.313-4.973)	Stevens & Neilson (1989)
Niagara R. Ft. Erie	1981	1.0		Fox et al. (1983)
	1986/87	0.82		NRDIG (1988)
	1988/89	1.19		NRDIG (1989)
Niagara-on-the-Lake, ON	1981/83	36	(9-310)	Oliver & Nicol (1984)
	1982	48		Oliver & Charlton (1984)
	1981	11		Fox et al. (1983)
	1986/87	2.48		NRDIG (1988)
	1988/89	2.11		NRDIG (1989)
Lake Ontario	1980	45	(33-64)	Oliver & Nicol (1982)
	1986	1.59	(0.554-3.784)	Stevens & Neilson (1989)
Grand R.	1980	10	(ND-42)	Oliver & Nicol (1982)

Table 4-3 Levels of 1,4-Dichlorobenzene in Surface Water in the United States

Media/Location	Year	Mean	(Range) min max	Reference
Estuarine Water (ng/L)				
Bayou d'Inde, Louisiana		74		Pereira et al. (1988)

Table 4-4 Levels of 1,4-Dichlorobenzene in Groundwater and Leachates in Canada

Media/Location	Year	Mean	(Range) min max	Reference
Ground water (ng/L)				
Municipal Landfill Sites				
North Bay, ON	1981		(7,000-40,000)	Reinhard et al. (1984)
Burlington, ON	1983		(ND-2,000)	Pankow et al. (1984a)
Burlington, ON	1983	170		Pankow et al. (1985)
Hazardous Waste Sites				
Ville Mercier, QC		400		Martel (1988)
Ville Mercier, QC	1988/89		(ND-3,800)	Martel & Ayotte (1989)
Napierville, QC	1987		(<2-7390 μg/l)	Foratek Intern. Inc. (1987)
Napierville, QC	1989		(<8-23100 μg/l) (63800 μg/L oil phase)	Foratek Inc., 1989

Table 4-5 Levels of 1,4-Dichlorobenzene in Soils, Sediment, and Sludge in Canada and the United States

Media/Location	Year	Mean min	(Range) max	Reference
Soil				
Napierville, QC	1987/89		(<2-2460 μg/g)	Foratek Intern. Inc. (1987)
Napierville, QC	1990		(<0.1-4.3μg/g)	ADS ass. Itée (1990)
Bottom (B) and Suspended (S) sediments (ng/g)				
Victoria, BC (B)	1991 (N=16)	141	(1-1710)	EVS Consultants (1992)
Great Lakes Basin				
urban runoff (S)	1979/83	40		Marsalek and Schroeter (1988)
Rainy R. ON (S)	1983	ND		Merriman (1987)
Lake Superior (B)	1980	5	(ND-9)	Oliver and Nicol (1982)
Lake Huron (B)	1980	16	(2-100)	Oliver and Nicol (1982)
Lake Erie (B)	1980	9	(3-20)	Oliver and Nicol (1982)
Niagara R.				
Niagara-on-the-Lake (S)	1979/81	180		Kuntz and Wary (1983)
Niagara R.				
Niagara-on-the-Lake (S)	1981		96-680	Fox et al. (1983)
Niagara R. (S)	1985/86		28 ng/L	Maguire et al. (1989)
Lake Ontario				
Niagara (S)	1982		(ND-160)	Oliver and Charlton (1984)
(B)	1982	63	(110-150)	Oliver and Charlton (1984)
(B)	1981		(33-1,300)	Fox et al. (1983)
(B)	1981		(ND-1,100)	Durham and Oliver (1983)
various sites (B)	1980	94	(22-210)	Oliver and Nicol (1982)

Table 4-5 cont'd   Levels of 1,4-Dichlorobenzene in Soils, Sediment,
                    and Sludge in Canada and the United States

Media/Location	Year	Mean (Range) min max	Reference
Bottom (B) and Suspended (S) sediments (ng/g)			
St. Lawrence R. (S)	1981	(1-27)	Merriman (1987)
(B)	1981	(1-9)	Merriman (1987)
(S)	1982/84	ND	Sylvestre (1987)
North Sask- (S)	1983	(ND-118)	Ongley et al. (1988)
atchewan R (B)	1983	(ND-107)	Ongley et al. (1988)
St.Clair R.ON (B)	1985	(ND-34,000)	Oliver & Pugsley (1986)
Economy R.,NS (B)	1979	(15-620)	MacLaren Marex Inc. (1979)
Salmon R., NS (B) Ind. outfall	1979	(92-440)	MacLaren Marex Inc. (1979)
Pictou Harbour, NS (B) Ind. outfall	1979	(37-1,000)	MacLaren Marex Inc. (1979)
St. Lawrence R. (S)	1981/87	2.7 ng/L	Germain & Langlois (1988)
Bayou d'Inde, Louisiana (B)		82000	Pereira et al. (1988)
Bayou d'Inde, Louisiana (S)		1200	Pereira et al. (1988)

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Table 4-5 cont'd   **Levels of 1,4-Dichlorobenzene in Soils, Sediment, and Sludge in Canada and the United States**

Media/Location	Year	Mean (Range) min max	Reference
Sewage Sludge (ng/g)			
Canadian	1980/85	(trace-1,500)	Webber and Lesage (1989)
WPCP Ontario	1986	(ND-2644)	OME (1988)

Table 4-6 Levels of 1,4-Dichlorobenzene in Biota in Canada

Media/Location	Year	Concentration Mean (Range) min. max.	Reference

Biota (ng/g wet weight)

Media/Location	Year	Concentration Mean (Range) min. max.		Reference
Oligochaetes Lake Ontario/ Niagara plume	1981	(ND-630)		Fox et al. (1983)
Amphipods Lake Ontario/ Niagara plume	1981	(ND-370)		Fox et al. (1983)
Lake Trout (Salvelinus namaycush)				
Lake Superior	1980	ND		Oliver and Nicol (1982)
Lake Huron	1980	1[1]		Oliver and Nicol (1982)
Lake Ontario	1980	3	(2-4)	Oliver and Nicol (1982)
	1981	ND		Fox et al. (1983)
Rainbow Trout (Salmo gairdneri)				
Lake Erie	1980	4[1]		Oliver and Nicol (1982)
Lake Ontario	1981	ND		Oliver and Niimi (1983)
Blue Mussels Pictou Harbour, NS at chemical plant	1979	(310-590)[2]		MacLaren Marex Ltd. (1979)

[1] only one sample analyzed
[2] total of 1,3- and 1,4-dichlorobenzene concentrations

Table 9-1 Acute Toxicity Studies of 1,4-Dichlorobenzene to Bacteria and Primary Producers

Species	Concentration Criterion	Concentration	Comments	References
Photobacterium phosphoreum	30 min. EC50 Microtox test	5.33 mg/L	50% light reduction	Ribo & Kaiser (1983)
Cyclotella meneghiniana	48-h EC50	34.30 mg/L	DNA reduction Static, unmeasured	Figueroa & Simmons (1991)
Selenastrum capricornutum	96-h EC50	1.6 mg/L	Growth inhibition static, measured	Calamari et et (1982)
Selenastrum capricornutum	3-h EC50	5.2 mg/L	Photosynthesis inhibition, static, measured	Calamari et al. (1983)
Selenastrum capricornutum	96-h EC50	1.6 mg/L	Growth inhibition unmeasured, static	Galassi & Vighi (1981)
Scenedesmus subspicatus	48-h LC50	38.0 mg/L	Growth inhibition unmeasured, static	Kuhn & Pattard (1990)
Scenedesmus pannonicus	48-h EC50	31.0 mg/L	Growth inhibition measured, static	Canton et al. (1985)
Ankistrodesmus falcatus falcatus	4-h EC50	19.9 mg/L (0.136 mmol)	Primary production Static, unmeasured	Wong et al. (1984)
Skeletonema costatum	48-h EC50	56.5 mg/L	Decrease of in vivo chlorophyll a	U.S. EPA (1978)

Table 9-2 Acute Toxicity Studies of 1,4-Dichlorobenzene to Aquatic Invertebrates

Species	Concentration Criterion	Concentration	Comments	References
Daphnia magna	48-h LC50	11.6 mg/L	Static, Unmeasured	Gersicht et al. (1986)
Daphnia magna	48-h EC50	11.0 mg/L	Static Unmeasured	US EPA (1978)
Daphnia magna	24-h IC50	1.6 mg/L	Immobilization, Static, measured	Calamari et al. (1983)
Daphnia magna	24-h IC50	1.6 mg/L	Immobilization, static, measured	Calamari et al. (1982)
Daphnia magna	48-h IC50	4.75 mg/L	Immobilization, Calculated	Hermens et al. (1984)
Artemia nauplii	24-h LC50	13.7 mg/L (93 mmol/m3)	Static, closed Unmeasured	Abernethy et al. (1986)
Daphnia magna	24-h LC50	42.0 mg/L	Static Unmeasured	LeBlanc (1980)
Daphnia magna	48-h LC50	11.0 mg/L	Static Unmeasured	LeBlanc (1980)
Daphnia magna	24-h EC50	3.2 mg/L	Reproduction delay Static, unmeasured	Kuhn et al. (1989)
Daphnia magna	48-h LC50	2.2 mg/L	Static Measured	Canton et al. (1985)

Table 9-2 cont'd   Acute Toxicity Studies of 1,4-Dichlorobenzene to Aquatic Invertebrates

Species	Concentration Criterion	Concentration	Comments	References
Daphnia magna	48-h EC50	0.74 mg/L	Static Measured	Canton et al. (1985)
Grass shrimp (Palaemonetes pugio)	96-h LC50	60.0 mg/L	Static Unmeasured	Curtis and Ward (1981)
Grass shrimp (Palaemonetes pugio)	48-h LC50	129.2 mg/L	Static Unmeasured	Curtis et al. (1979)
Grass shrimp (Palaemonetes pugio)	96-h LC50	69.0 mg/L	Static Unmeasured	Curtis et al. (1979)
Midge (Tanytarsus dissimilis)	48-h LC50	13.0 mg/L	Static Measured	Call et al. (1985)

Table 9-3 Chronic Toxicity Studies of 1,4-Dichlorobenzene to Aquatic Invertebrates

Species	Concentration Criterion	Concentration	Comments	References
Daphnia magna	21-day NOEC	0.50 mg/L	Reproduction Static, measured	Kuhn et al. (1989)
Daphnia magna	16-day LC50	1.5 mg/L (log 1.01)	Mortality Calculated	Hermens et al. (1984)
Daphnia magna	16-day EC50	0.47 mg/L (log 0.51)	Reproduction Calculated	Hermens et al. (1984)
Daphnia magna	16-day NOEC	0.58 mg/L (log 0.60)	Growth Calculated	Hermens et al. (1985)
Daphnia magna	16-day EC50	0.68 mg/L (log 0.67)	Reproduction Calculated	De Wolf et al. (1988)
Daphnia magna	28-day LOEC	0.40 mg/L	Fertility impaired measured	Calamari et al. (1982)
Daphnia magna	28-day NOEC	0.22 mg/L	Reproduction measured	Calamari et al. (1982)
Daphnia magna	14-Day EC50	0.93 mg/L	Reproduction measured	Calamari et al. (1983)

Table 9-4   Acute Toxicity Studies of 1,4-Dichlorobenzene to Fish

Species	Concentration Criterion	Concentration	Comments	References
Fathead minnows (Pimephales promelas)	96-h LC50	4.2 mg/L	Flow through Measured	Carlson & Kosian (1987)
Fathead minnow (Pimephales promelas)	96-h LC50	4.16 mg/L	Flow through Measured	Ahmad et al. (1984)
Fathead minnows (Pimephales promelas)	96-h LC50	4.0 mg/L	Flow through Measured	US EPA (1980)
Fathead minnows (Pimephales promelas)	96-h LC50	4.0 mg/L	Flow through Measured	Veith et al. (1983)
Fathead minnows (Pimephales promelas)	96-h LC50	30.0 mg/L	Static Unmeasured (H)	Curtis and Ward (1981)
Fathead minnows (Pimephales promelas)	96-h LC50	33.7 mg/L	Flow through Measured	Curtis et al. (1979)
Fathead minnows (Pimephales promelas)	96-h LC50 (Fry: 10-15 days)	3.6 mg/L	Static Unmeasured	Mayes et al. (1983)
Fathead minnows (Pimephales promelas)	96-h LC50 (Juvenile: 30-35 days)	14.2 mg/L	Static Unmeasured	Mayes et al. (1983)
Fathead minnows (Pimephales promelas)	96-h LC50 (Subadult: 60-100 days)	11.7 mg/L	Static Unmeasured	Mayes et al. (1983)

Table 9-4 cont'd   Acute Toxicity Studies of 1,4-Dichlorobenzene to Fish

Species	Concentration Criterion	Concentration	Comments	References
Rainbow trout (Oncorhynchus mykiss)	24-h LC50	1.2 mg/L	Static Measured	Calamari et al. (1982)
Rainbow trout (Oncorhynchus mykiss)	48-h LC50	1.18 mg/L	Static Measured	Calamari et al. (1983)
Rainbow trout (Oncorhynchus mykiss)	96-h EC50	1.10 mg/L	Flow through Measured	Ahmad et al. (1984)
Rainbow trout (Oncorhynchus mykiss)	96-h LC50	1.12 mg/L	Flow through Measured	Ahmad et al. (1984)
Rainbow trout (Oncorhynchus mykiss)	96-h LC50	1.12 mg/L	Flow through Measured	US EPA (1980)
Rainbow trout (Oncorhynchus mykiss)	48-h LD50	1.48 mg/L	Flow through Measured	US EPA (1980)
Rainbow trout (Oncorhynchus mykiss)	48-h LC50	1.24 mg/L	Flow through Measured	US EPA (1983)
Rainbow trout (Oncorhynchus mykiss)	96-h LC50	1.12 mg/L	Flow through Measured	US EPA (1983)
Sheepshead minnows (Cyprinodon variegatus)	48-h LC50 96-h LC50	7.2 mg/L 7.4 mg/L	Static Unmeasured	Heitmuller et al. (1981)

Table 9-4 cont'd   Acute Toxicity Studies of 1,4-Dichlorobenzene to Fish

Species	Concentration Criterion	Concentration	Comments	References
Zebrafish (Brachydanio rerio)	24-h LC50	4.2 mg/L	Static Measured	Calamari et al. (1982)
Zebra fish (Brachydanio rerio)	48-h LC50	4.25 mg/L	Static Measured	Calamari et al. (1983)
Zebra fish (Brachydanio rerio)	48-h LC50	10.0 mg/L	Flow through Unmeasured	Sloof (1979)
American flagfish (Jordanella floridae)	96-h LC50	2.45 mg/L	1 Week old Flow through, unmeasured	ATRG (1987)
American flagfish (Jordanella floridae)	96-h LC50	2.05 mg/L	2-4 months Flow through, unmeasured	ATRG (1987)

Table 9-5 Chronic Toxicity Studies of 1,4-Dichlorobenzene to Fish

Species	Concentration Criterion	Concentration	Comments	References
Rainbow trout (Oncorhynchus mykiss)	14 day LC50	0.80 mg/L	Static, measured	Calamari et al. (1982)
Fathead minnow (Pimephales promelas)	32 day exposure	8.7 ±0.78 mg/L (8,700 µg/L)	Exposure to embryos caused deformities.	Carlson & Kosian (1987)
Fathead minnow (Pimephales promelas)	32 day NOEC	0.57 ±0.67 mg/L	Flow through measured	Carlson & Kosian (1987)
Fathead minnow (Pimephales promelas)	32 day LOEC	1.0 mg/L	Flow through measured	Carlson & Kosian (1987)
Fathead minnow (Pimephales promelas)	32-day MATC	0.565 to 10.4 0.565 mg/L 1.04 mg/L	Flow through measured	Ahmad et al. (1984)
Fathead minnow (Pimephales promelas)	7 day exposure	2.8 mg/L	MATC effect growth Static, unmeasured	Mayes et al. (1988)
Guppy (Poecilia reticulata)	14 day LC50	3.95 mg/L (log 1.43)	Static Unmeasured	Konemann (1981)
American flagfish (Jordanella floridae)	28-d MATC	>0.349 mg/L	fry survival & growth	ATRG (1987)

Table 9-5 cont'd Chronic Toxicity Studies of 1,4-Dichlorobenzene to Fish

| Species	Concentration Criterion	Concentration	Comments	References
Zebra fish (Brachydanio rerio)	28 day LC50	2.7 mg/L	Semistatic measured	van Leeuwen et al. (1990)
Zebra fish (Brachydanio rerio)	28 day NOLC	2.1 mg/L	Semistatic measured	van Leeuwen et al. (1990)
Zebra fish (Brachydanio rerio)	28 day NOEC	0.65 mg/L	Growth Semistatic,measured	van Leeuwen et al. (1990)

[a] MATC - maximum acceptable toxicant concentration

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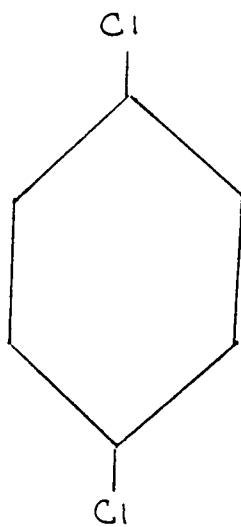
Figure 1. Structural formula of 1,4-dichlorobenzene