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detected in some environmental media (HazDat 2005). 1,4-DCB has been identified in air samples collected at 23 of the 330 NPL hazardous waste sites where it has been detected in some environmental media (HazDat 2005).

6.2.2 Water

According to the TRI, the estimated releases of 1,2-DCB of 1,240 pounds (0.6 metric tons) to water from 39 large processing facilities accounted for 1% of the total TRI environmental releases in 2003 (TRI03 2005). An additional 1,104 pounds (0.5 metric tons) (1% of total TRI environmental releases) were released off-site, which includes release to publicly owned treatment works (POTWs). Table 6-1 lists amounts of 1,2-DCB released from these facilities. Estimated releases of 1,3-DCB of 363 pounds (0.2 metric tons) to water from eight large processing facilities accounted for 18% of the total TRI environmental releases in 2003 (TRI03 2005). An additional 255 pounds (0.1 metric tons) (13% of total TRI environmental releases) were released off-site, which includes release to POTWs. Table 6-2 lists amounts of 1,3-DCB released from these facilities. Estimated releases of 1,4-DCB of 815 pounds (0.4 metric tons) to water from 21 large processing facilities accounted for 0.8% of the total TRI environmental releases in 2003 (TRI03 2005). An additional 297 pounds (0.1 metric tons) (0.3% of total TRI environmental releases) were released off-site, which includes release to POTWs. Table 6-3 lists amounts of 1,4-DCB released from these facilities. The TRI data should be used with caution because only certain types of facilities are required to report (EPA 1997b). Therefore, this is not an exhaustive list.

Less than 1% of environmental releases of 1,4-DCB are to surface water (EPA 1981a). The main route for the release of this substance to surface water is expected to be through its extensive use in urinal deodorant blocks (IARC 1999). 1,2-DCB is released into industrial waste water during its production and use. 1,2-DCB might also be released into waste water during the disposal of this substance when it is produced as a by-product in the production of 1,4-DCB. Data concerning the release of 1,3-DCB to water are lacking. Release of this substance to water may occur during its production, use, or disposal. DCBs have been identified in industrial and municipal waste waters from several sources, at concentrations ranging from <3 to >900 ppb (Oliver and Nichol 1982a; Perry et al. 1979; Young and Heesen 1978; Young et al. 1980, 1981). 1,2- and 1,4-DCB were both detected in 1% of 84 possible detections in influent samples from the New York City municipal waste water treatment system at concentrations of 22 and 4 ppb, respectively (Stubin et al. 1996). 1,2-DCB was detected in 2% while 1,4-DCB was detected in 1% of 84 possible detections in effluent samples at concentrations of 4–6 and 3 ppb,

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respectively. The concentrations of 1,2-DCB were higher than those of 1,4-DCB, which is contrary to what is expected for these substances in residential and domestic waste water. However, no explanation was offered for this. The concentration of 1,4-DCB in the effluent of the North Regional Wastewater Treatment Plant in Broward County, Florida was approximately 1.2 ppb (Tansel and Eyma 1999). 1,4-DCB was detected above “standard levels” (unspecified) in sediment at the end of the Macaulay Point and Clover Point waste water outfalls off the coast of Vancouver, British Columbia (Taylor et al. 1998).

DCB (unspecified isomers) has been reported in the leachate from industrial and municipal landfills at concentrations from 0.007 to 0.52 ppm (7–520 ppb) (Brown and Donnelly 1988). Eganhouse et al. (2001) identified 1,4-DCB at a concentration of 0.1–5.6 ppb in a landfill leachate plume in groundwater from a municipal landfill located in Norman, Oklahoma. DCBs have also been detected in wetland-treated leachate water at a municipal solid waste landfill in central Florida (Chen and Zoltek 1995). Groundwater samples contained 1,2-DCB at concentrations of 0.09–1.56 ppb, 1,3-DCB at concentrations of 0.08–8.95 ppb, and 1,4-DCB at concentrations of 0.08–10.71 ppb. Hallbourg et al. (1992) detected DCB (unspecified isomers) in groundwater at several landfill sites in Orange County, Florida. These authors reported mean concentrations of DCBs of 0.37–21.2, 6–46.4, and <1–7.4 ppb at the Orange County Landfill, Alachua County Southwest Landfill, and the Alachua County Northeast Landfill, respectively. In their study, DCB was one of the 10 most frequently detected volatile organic compounds (VOCs). Plumb (1991) also reported 1,2-, 1,3-, and 1,4-DCB in groundwater samples collected at 36, 16, and 34 of 479 hazardous waste sites, respectively.

1,4-DCB was monitored for, but not detected, in 86 samples of urban storm water runoff in the National Urban Runoff Program (Cole et al. 1984). DCBs were detected in four rivers (Aire, Calder, Don, and Trent) that drain an industrial catchment from the United Kingdom into the North Sea (Meharg et al. 2000). Annual fluxes in these rivers ranged from 1.37 to 32.91 kg/year for 1,2-DCB, 0.12 to 9.33 kg/year for 1,3-DCB, and 6.80 to 28.96 kg/year for 1,4-DCB.

1,2-DCB has been identified in surface water and groundwater samples collected at 29 and 186 of the 281 NPL hazardous waste sites, respectively, where it was detected in at least one environmental medium (HazDat 2005). 1,3-DCB has been identified in surface water and groundwater samples collected at 13 and 107 of the 175 NPL hazardous waste sites, respectively, where it was detected in some environmental media (HazDat 2005). 1,4-DCB has been identified in surface water and groundwater samples collected at 31 and 213 of the 330 NPL hazardous waste sites, respectively, where it was detected in some environmental media (HazDat 2005).

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6.2.3 Soil

According to the TRI, releases of 1,2-DCB to land of 1,784 pounds (0.8 metric tons) from 39 large processing facilities accounted for 2% of total TRI environmental releases in 2003 (TRI03 2005). An estimated 2,500 pounds (1 metric ton) (3% of total TRI environmental releases) were released via underground injection. Table 6-1 lists amounts of 1,2-DCB released from these facilities. Releases of 1,3-DCB of 255 pounds (0.1 metric tons) to the land from eight large processing facilities accounted for 13% of total TRI environmental releases in 2003 (TRI03 2005). Table 6-2 lists amounts of 1,3-DCB released from these facilities. There were no releases of 1,3-DCB to the underground in 2003 as shown in Table 6-2. Releases of 1,4-DCB of 270 pounds (0.1 metric tons) to the land from 21 large processing facilities accounted for 0.2% of total TRI environmental releases in 2003 (TRI03 2005). In addition, an estimated 10,408 pounds (0.5 metric tons) (11% of total environmental releases) were released via underground injection. Table 6-3 lists amounts of 1,4-DCB released from these facilities. The TRI data should be used with caution because only certain types of facilities are required to report (EPA 1997b). Therefore, this is not an exhaustive list.

The principal sources of 1,4-DCB release to land are disposal of industrial waste in landfills, application of sewage sludge containing 1,4-DCB to agricultural land, and atmospheric deposition (Wang and Jones 1994b; Wang et al. 1995). Municipal wastes may include unused space deodorants and moth repellents containing 1,4-DCB, but these releases are not expected to be significant (EPA 1981a). A survey of 204 sewage sludges conducted in Michigan that analyzed for 73 organic compounds reported a concentration range of 0.04–633 mg/kg dry weight (ppm) for 1,4-DCB and mean and median concentrations of 12.0 and 2.02 ppm, respectively (Jacobs and Zabik 1983). 1,4-DCB from this source may be released to soils during land applications of sludge to agricultural soils. A similar survey of sewage sludges in England found 1,4-DCB ranging from 561 to 2,320 µg/kg (0.561–2.32 ppm wet weight) in all 12 of the samples tested (Wang and Jones 1994b). Wang et al. (1995) reported, however, that 1,4-DCB concentrations increased during the 1960s in both plots receiving sewage sludge applications and in control soil plots. The authors concluded that atmospheric deposition during the 1960s in particular, which corresponded to a period of increased production of many organochlorine compounds, was a likely source. 1,2-DCB was detected in all 12 sewage sludge samples at concentrations ranging from 71.3 to 4,110 µg/kg (ppb) dry weight (3.57–152 ppb wet weight). The concentrations of 1,2-DCB in industrial sewage sludge was considerably higher than in urban sewage

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sludge. 1,3-DCB was detected in 9 out of 12 sewage sludge samples at concentrations ranging from below the detection limit to 467 µg/kg (ppb) dry weight (below the detection limit–13.5 ppb wet weight).

1,2-DCB is produced in large quantities as a by-product in the production of 1,4-DCB. The TRI data for this substance suggest that 1,2-DCB may be released into the ground during the disposal of unused supplies. Data concerning the release of 1,3-DCB to soil were lacking. Based on TRI data, the production volume of these chemicals, and their uses, releases of this isomer to soil are expected to be minor compared to the other DCB isomers.

1,2-DCB has been identified in soil and sediment samples collected at 111 and 37 of the 281 NPL hazardous waste sites, respectively, where it was detected in at least one environmental medium (HazDat 2005). 1,3-DCB has been identified in soil and sediment samples collected at 64 and 25 of the 175 NPL hazardous waste sites, respectively, where it was detected in at least one environmental medium (HazDat 2005). 1,4-DCB has been identified in soil and sediment samples collected at 112 and 52 of the 330 NPL hazardous waste sites, respectively, where it was detected in at least one environmental medium (HazDat 2005).

6.3 ENVIRONMENTAL FATE

6.3.1 Transport and Partitioning

Whereas 1,2- and 1,3-DCB are liquids at room temperature, 1,4-DCB is a solid that sublimates readily. Sublimation rates of 1,4-DCB from consumer products were measured at 1.6×10^{-3} to 4.6×10^{-3} g/minute at temperatures ranging from 21 to 24 °C during a 19-day test period (Scuderi 1986). DCBs tend to volatilize to the atmosphere from soil and water at a relatively rapid rate. The estimated volatilization half-life for these chemicals was 4 hours in a model river and 120 hours from a model lake (HSDB 2005). The reported volatilization half-lives for 1,4-DCB measured in coastal seawater ranged from 10 to 18 days (Wakeham et al. 1983). 1,2-DCB (100 ppm) and 1,4-DCB (300 ppm) both volatilized completely from nonaerated distilled water in <3 days and from aerated distilled water in <4 hours (Garrison and Hill 1972). Volatilization from surface soil may be an important transport mechanism for DCBs (Wang and Jones 1994a), but adsorption to soil particulates may inhibit volatilization (Wilson et al. 1981).

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Since DCBs are slightly soluble in water (80.0–156 mg/L) (Banerjee et al. 1980; Miller et al. 1984; Yalkowsky and He 2003), partitioning to clouds, rain, or surface water may occur. Henry's Law constant values ranging from 1.74×10^{-3} to 2.63×10^{-3} atm-m³/mol at 25 °C (Shiu and Mackay 1997; Staudinger and Roberts 1996) indicate that partitioning from air to water is likely to be minor relative to the reverse process of volatilization of the compound from water to air. However, DCBs have been detected in rainwater and snow (Laniewski et al. 1998, 1999; Ligocki et al. 1985). The concentration of 1,4-DCB detected in 6 of 7 rainwater samples collected in Portland, Oregon, ranged from 3 to 7 ppt (ng/L), while the concentration of 1,2-DCB detected in 5 out of 7 rainwater samples ranged from 0.13 to 0.62 ppt (Ligocki et al. 1985). DCBs have been detected in surficial snow from Antarctica (Laniewski et al. 1998), which suggests that these substances can be transported over long distances through the atmosphere.

Based on measured soil organic carbon partition coefficient (K_{oc}) values, which range from 275 to 1,833 in different soils (Bahnick and Doucette 1988; Chiou et al. 1983; Newsom 1985; Schwartzbach and Westall 1981; Wilson et al. 1981), DCBs are expected to sorb moderately to soils and sediments. Sorption is primarily to the soil organic phase (Chiou et al. 1983) and, therefore, depends on the organic content of the soil. However, sorption is likely to be reversible; therefore, DCBs may leach from hazardous waste sites and be transported to groundwater, or may migrate from surface water through the soil to groundwater (Newsom 1985; Schwartzbach and Westall 1981). In a sandy soil with low organic matter, 26–49% of 1,4-DCB percolated through the soil to a depth of 140 cm (Wilson et al. 1981).

DCBs are expected to bioconcentrate in aquatic organisms. High log octanol-water partition coefficient ($\log K_{ow}$) values of 3.43–3.53 (Hansch et al. 1995) also suggest that DCBs have a moderate to high potential for bioaccumulation. A calculated bioconcentration factor (BCF) of 267 was reported for the fathead minnow (*Pimephales promelas*) (ASTER 1995). Measured mean BCF values of 370 and 720 were experimentally determined at equilibrium for rainbow trout exposed to water concentrations of 28 ng/L (ppb) and 670 ng/L (ppb), respectively, of 1,4-DCB for up to 119 days in laboratory aquaria (Oliver and Niimi 1983). BCF values measured in this study for 1,2-DCB were 270 (47 ng/L in water) and 560 (940 ng/L in water), while BCF values measured for 1,3-DCB were 420 (28 ng/L in water) and 740 (690 ng/L in water). A study of chlorobenzenes in sediments, water, and selected fish from the Great Lakes indicated that many chlorobenzenes are bioconcentrated by fish, but that DCBs are concentrated to a smaller extent than some of the more highly chlorinated chlorobenzene compounds such as pentachlorobenzene and hexachlorobenzene (Oliver and Niimi 1982a). For example, equilibrium/steady-

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state BCF values measured in fish maintained in flowing water systems typically increased with increasing chlorination as shown in Table 6-4.

DCBs can enter soil-plant systems through many routes including atmospheric deposition, sewage sludge application to agricultural land, and through industrial activities (Wang and Jones 1994a). Wang and Jones (1994c) studied the uptake of several chlorobenzene compounds in carrots grown in spiked and sewage-amended soils. The transfer of chlorobenzenes from soils to plants and subsequent bioaccumulation is of interest because chlorobenzenes are ubiquitous in sewage sludge. Chlorobenzenes are also lipophilic and volatile compounds that can be taken up by plants by both root and foliage pathways. Carrots were grown for 100 days in control soil, chemically-spiked soil, and in low and high rate sludge-amended soils. DCB concentration in the soils did not remain constant throughout the growth period. BCF values are not traditional steady-state values since measurements were taken for only one time interval. The authors reported that concentrations of 1,4-DCB in soil before sowing and after the harvest were 5.9 and 2.6 ppb dry weight in the control, 16 and 11 ppb in the chemically-spiked soil, 10 and 7.4 ppb in the low rate sewage-amended soil, and 38 and 30 ppb in the high rate sewage-amended soils, respectively. Concentrations of 1,4-DCB in carrot foliage and the corresponding bioconcentration factors (BCFs) were 13 ppb (BCF=3.1) for the control, 17 ppb (BCF=1.3) for the spiked soil, 22 ppb (BCF=2.5) for the low rate sewage-amended soil, and 49 ppb (BCF=1.5) for the high rate sewage-amended soil. The concentrations of 1,2-DCB in soil before sowing and after the harvest were both below the detection limit (unspecified) in the control, 29 and 17 ppb in the chemically-spiked soil, 13 and 7.3 ppb in the low rate sewage-amended soil, and 60 and 45 ppb in the high rate sewage-amended soils, respectively. Concentrations of 1,2-DCB in carrot foliage and the corresponding BCFs were 6.7 ppb (BCF not given) for the control, 9.6 ppb (BCF=0.42) for the spiked soil, 12 ppb (BCF=1.2) for the low rate sewage-amended soil, and 26 ppb (BCF=0.49) for the high rate sewage-amended soil. The concentrations of 1,3-DCB in soil before sowing and after the harvest were both below the detection limit (unspecified) in the control, 4.2 and 2.9 ppb in the chemically-spiked soil, 2.3 and 0.98 ppb in the low rate sewage-amended soil, and 8.2 and 5.8 ppb in the high rate sewage-amended soils, respectively. Concentrations of 1,3-DCB in carrot foliage and the corresponding BCFs were 0.72 ppb (BCF not given) for the control, 0.83 ppb (BCF=0.24) for the spiked soil, 1.3 ppb (BCF=0.80) for the low rate sewage-amended soil, and 2.2 ppb (BCF=0.31) for the high rate sewage-amended soil. The application of the low-rate sewage sludge stimulated both the carrot foliage and root production to the greatest extent. The authors concluded that foliar uptake of all chlorobenzenes tested, including the DCBs, was an important bioaccumulation pathway.

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Table 6-4. Comparison of Bioconcentration Factors (BCFs) for Various Chlorinated Benzenes in Fish

Compound	BCF (range)
Monochlorobenzene	12–450
1,2-Dichlorobenzene	89–560
1,3-Dichlorobenzene	66–740
1,4-Dichlorobenzene	15–720
1,2,3-Trichlorobenzene	700–2,600
1,2,4-Trichlorobenzene	182–3,200
1,3,5-Trichlorobenzene	760–4,100
1,2,3,4-Tetrachlorobenzene	3,800–12,000
1,2,3,5-Tetrachlorobenzene	1,800–3,900
1,2,4,5-Tetrachlorobenzene	4,000–13,000
Pentachlorobenzene	3,400–20,000
Hexachlorobenzene	12,000–44,437

Source: EPA 1985a

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The concentrations (dry weight) of the DCBs in the carrot peel were typically equal to or slightly lower than the concentrations in the carrot core (Wang and Jones 1994a). This indicated that DCBs, when present in carrots, penetrate into the core. For carrot roots, the concentrations of 1,4-DCB in the core and peel were 9.4 µg/kg (ppb) (BCF=2.2) and 7.0 ppb (BCF=1.6) for the control, 5.9 ppb (BCF=0.44) and 7.3 ppb (BCF=0.54) for the chemically-spiked soil, 5.9 ppb (BCF=0.68) and 5.8 ppb (BCF=0.67) for the low-rate sewage application, and 9.6 ppb (BCF=0.28) and 4.3 ppb (BCF=0.13) for the high-rate sewage treatment, respectively. The concentrations of 1,2-DCB in the core and peel were 1.5 µg/kg (ppb) (BCF not given) and 1.4 ppb (BCF not given) for the control, 5.8 ppb (BCF=0.25) and 5.3 ppb (BCF=0.23) for the chemically-spiked soil, 0.0 ppb (BCF=0.0) and 0.84 ppb (BCF=0.085) for the low-rate sewage application, and 2.8 ppb (BCF=0.053) and 1.5 ppb (BCF=0.029) for the high-rate sewage treatment, respectively. 1,3-DCB was only detected in the core of the chemically-spiked soil at 1.0 ppb (BCF=0.29) and in the core of the high-rate sewage treatment at 1.8 ppb (BCF=0.26). 1,3-DCB concentrations in the root peels as well as the root core of the control were below the detection limit (unspecified). Overall, <1% of the DCBs and other chlorobenzenes in the soil were accumulated by the carrots, which is minor compared with the other loss pathway from the soil, principally volatilization.

Wang et al. (1996) found that a 1 ppm solution of 1,4-DCB was taken up by carrots (*Daucus carota*, 49%), soybeans (*Glycine max*, 50%), and red goosefoot (*Chenopodium rubrum*, 62%), but not by tomatoes (*Lycopersicon esculentum*). Only the soybean cell cultures provided evidence of the existence of metabolites of this compound, probably conjugates of chlorophenol. The authors further observed that the uptake, metabolism, and toxicity of 1,4-DCB differed among the species tested.

Zhang et al. (2005) studied DCB uptake in vegetables grown in urban areas of China. DCB concentrations in spinach, Chinese cabbage, and celery were highest in roots, followed by leaves. Concentrations in radishes and carrots were highest in leaves, followed by stems. The authors reported that the accumulation of chlorinated benzenes in these vegetables was affected by the lipid contents of the vegetables, the volatilities of the chemicals, and the physiological characteristics of the vegetables.

Data on biomagnification of DCBs through aquatic or terrestrial food chains were not located.

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6.3.2 Transformation and Degradation**6.3.2.1 Air**

The main degradation pathway for DCBs in air is reaction with photochemically generated hydroxyl radicals (Cuppitt 1980; EPA 1985a). Reactions with ozone or other common atmospheric species are not expected to be significant (Cuppitt 1980; EPA 1985d). Therefore, the atmospheric lifetime of the DCBs may be predicted from an assumed hydroxyl radical concentration in air and the rate of reaction. The reported rate for reaction of hydroxyl radicals with DCBs is $3.2\text{--}7.2 \times 10^{-13}$ cm³/mol-sec (Atkinson 1989; Howard 1989), and the estimated atmospheric half-life for DCBs is about 14–31 days (Howard 1989). Since this degradation process is relatively slow, DCBs may become widely dispersed, but are not likely to accumulate in the atmosphere. The degradation pathways for 1,4-DCB in the atmosphere are shown in Figure 6-4.

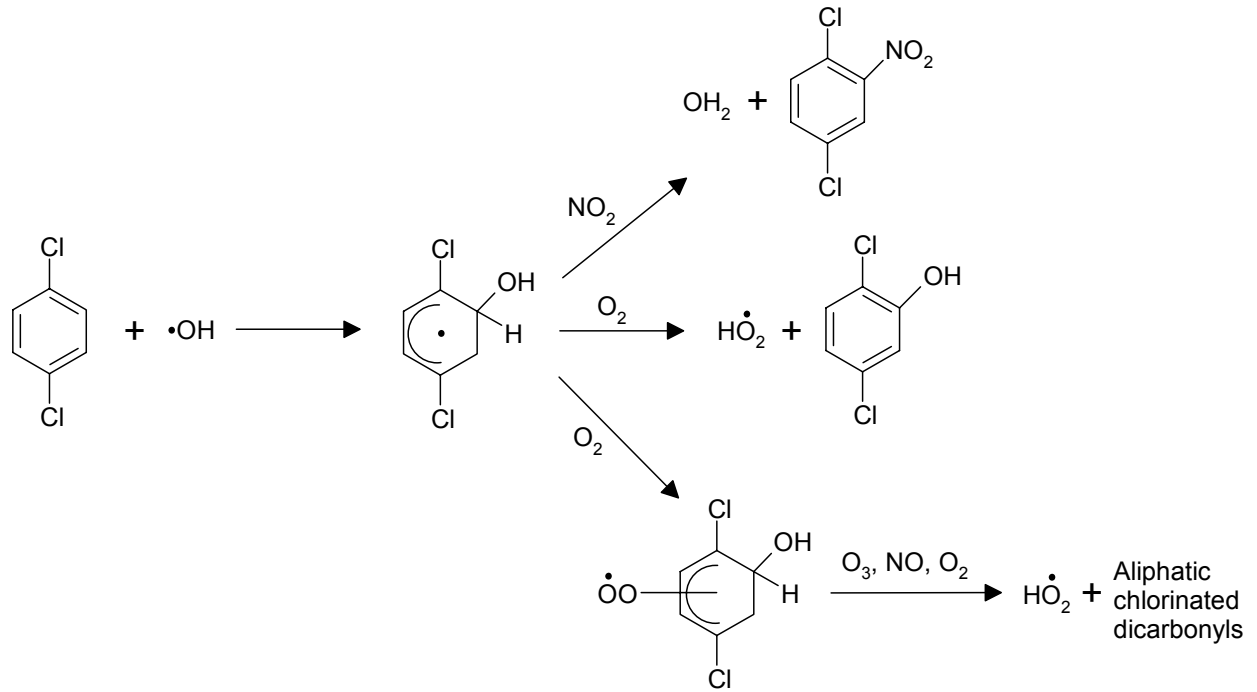
Reports of smog chamber studies of chlorobenzene degradation have indicated degradation after 5 hours of 21.5% of 1,2-DCB (EPA 1985a). Chloronitrobenzenes and chloronitrophenols were identified as degradation products. Irradiation of chlorobenzenes with natural sunlight was reported to produce polychlorinated biphenyls (PCBs). Whether this occurs under natural atmospheric conditions is unknown, but it would appear to be unlikely because of the normally low concentrations of chlorobenzenes in ambient air.

6.3.2.2 Water

Biodegradation may be an important transformation process for DCBs in water under aerobic, but not anaerobic, conditions (Bouwer and McCarty 1982, 1983, 1984; Schwartzenbach et al. 1983; Spain and Nishino 1987; Tabak et al. 1981). Although volatilization of 1,4-DCB may interfere with biodegradation studies, ¹⁴C studies indicate that significant biodegradation of 1,4-DCB does occur (Spain and Nishino 1987). Longer acclimation periods are required when 1,4-DCB is the sole carbon source (Spain and Nishino 1987).

Several aerobic screening tests have been performed on the DCB isomers. 1,2- and 1,3-DCB, both at initial concentrations of 30 mg/L, reached 0% of their theoretical BOD in 4 weeks using an activated sludge inoculum at 100 mg/L and the Japanese MITI test (CITI 1992). During an OECD closed bottle test, removal of 1,4-DCB was 97.1%. However, volatilization was considered to be the major mechanism

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Figure 6-4. The Decomposition of 1,4-Dichlorobenzene in Air

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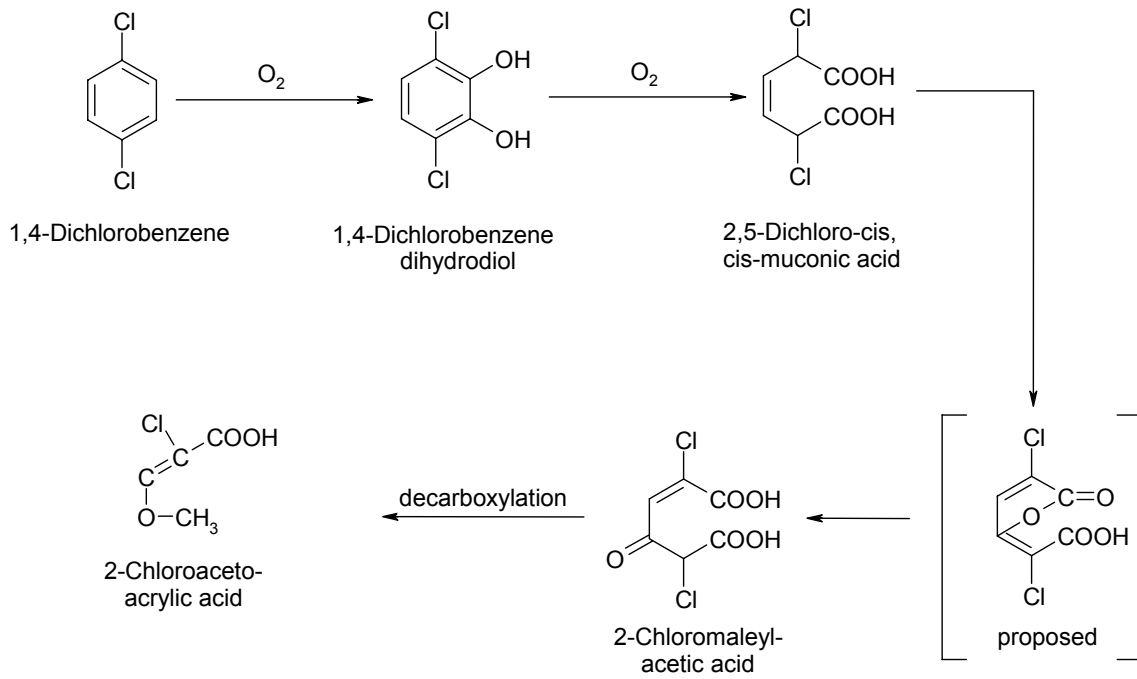
for removal. During a modified porous pot test operated under normal conditions at a lower aeration rate, temperatures of 8, 15, and 20 °C, and sludge retention times of 3 and 6 days, removal of 1,4-DCB was >95%. The author reported that the major mechanism for 1,4-DCB removal in this test was biodegradation. Using acetate as the primary carbon source under aerobic conditions and after an acclimation period of 10 days, rapid bacterial degradation of 96% of a 1,2-DCB sample, 28% of a 1,3-DCB sample, and 98% of a 1,4-DCB sample was reported (Bouwer and McCarty 1982). 1,4-DCB was completely mineralized to inorganic end products. Possible explanations for the lower 1,3-DCB biodegradation rate were biodegradation with slow utilization kinetics or sorption removal. The biodegradation rate of 1,2-DCB in a heterogeneous unconfined aquifer at Columbus Air Force Base in Columbus, Mississippi was measured to be 0.0059 day⁻¹ (Stauffer et al. 1994). This corresponds to a half-life of 117 days. Biodegradation of 1,2-DCB in aquifer samples from Vejen and Grindsted, Denmark was slow, with >30% of the test compound remaining after 50 days. 1,4-DCB was not degraded in these samples after 50 days. 1,2-DCB (initial concentrations, 20 ppm) underwent 30–50% biodegradation in river water and 15–30% biodegradation in sea water after 3 days during an aerobic screening test (Kondo et al. 1988). 1,4-DCB (initial concentrations, 4 ppm) underwent 0% biodegradation in both the river water and sea water inocula after 3 days. *In-situ* biodegradation rate constants were measured for 1,2- and 1,4-DCB in an aerobic aquifer (Nielsen et al. 1996). Rate constants and lag phases were 0.02–0.06 day⁻¹ (half-life, 12–35 days) and 0–20 days, respectively, for 1,2-DCB and 0.01–0.05 day⁻¹ (half-life, 14–69 days) and 0–22 days, respectively, for 1,4-DCB. Half-lives reported for 1,4-DCB in seawater mesocosm experiments performed at various temperatures ranged from 10 to 18 days (Wakeham et al. 1983). The authors noted that volatilization was the dominant removal process. No degradation of DCBs was reported under denitrification or methanogenic conditions (Bouwer and McCarty 1983, 1984). Degradation pathways for 1,4-DCB in water are shown in Figure 6-5.

6.3.2.3 Sediment and Soil

Based on the Henry's law constants of 1,2- and 1,3-DCB and the tendency of 1,4-DCB to sublime, volatilization rather than transformation is the most likely fate process for DCBs from surface soil. Transformation of DCBs by biodegradation, photolysis, chemical hydrolysis, and oxidation appear to be relatively minor processes. Leaching of DCBs to groundwater from subsurface soils under certain conditions may occur (EPA 1985a).

Wang and Jones (1994a) studied the fate of chlorobenzenes including DCBs in chemically-spiked and sewage-amended soils to determine the rate of volatilization, biodegradation, photolysis, and other

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Figure 6-5. The Decomposition of 1,4-Dichlorobenzene in Soil and Water

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possible loss processes. These authors used sewage sludge collected from a sewage treatment facility serving a municipal (~60%) and industrial (~40%) catchment. The sewage sludge or chemically-spiked solutions containing chlorobenzenes were added to five experimental systems; (1) normal soil, (2) sterilized soil (with 1% [weight] of sodium azide), (3) sterilized soil shaded with aluminum foil, (4) sterilized soil, shaded and sealed with a Teflon-coated septum, and (5) a control (untreated soil). The mesocosms were incubated at 20–30 °C over a 259-day period. Loss of all chlorobenzenes including DCBs was best represented by a two-step first-order kinetics model. In the normal condition containing unsterilized soil exposed to sunlight and open to the air, during the first 35 days, 79.9% of the 1,2-DCB, 85.1% of the 1,3-DCB, and 70.5% of the 1,4-DCB were lost with half-life values of 13.2, 12.4, and 17.4 days, respectively. From day 35 to day 259, only 4.29% of 1,2-DCB, 3.93% of 1,3-DCB, and 11.3% of 1,4-DCB were lost with half-life values of 892, 579, and 294 days, respectively. For the chemically-spiked soil treatment, the first phase (days 0–17) loss was 75.6% for 1,2-DCB, 73.3% for 1,3-DCB, and 73.2% for 1,4-DCB with half-life values of 8.63, 8.42, and 8.57 days. The second phase (days 17–259) loss was 13.9% for 1,2-DCB, 25.4% for 1,3-DCB, and 11.2% for 1,4-DCB with half-lives of 191, 189, and 131 days, respectively. Although the DCB loss rates in the sewage-amended soil were slower than those in the chemically-spiked soil, the total percentage losses of DCBs after 259 days were comparable. Based on the results of losses of DCBs observed in the other microcosm systems, the authors concluded that transformation processes including biodegradation, photolysis, and other abiotic losses (chemical hydrolysis and oxidation) were minor processes compared to volatilization. The experimental results of Wang and Jones (1994a) showed that, during the first phase, volatilization rates were high and substantial portions of the DCBs were lost. The second phase was much slower and portions of the DCBs remained in the soil for a much longer period.

Neither 1,3- nor 1,4-DCB were biotransformed in an aerobic Rhine River sediment column (closed system) after 12 months (Bosma et al. 1990). 1,2-DCB was completely degraded after 4 months following a lag period of 60–100 days. DCBs (unspecified isomers) were degraded slowly in alkaline para-brown soil (100 g soil per 2 mg compound) with 6.3% of theoretical CO₂ evolution in a closed system after 10 weeks (Haider et al. 1974). Half-lives corresponding to the biodegradation of 1,2-, 1,3-, and 1,4-DCB in anaerobic estuarine sediment from the Tsurumi River, Japan were 36.9, 433.2, and 385.1 days, respectively (Masunaga et al. 1996). Between 25 and 90% of 1,2- and 1,4-DCB were removed from an aerobic soil column (closed system) after 300 days of continuous operation, while <25% of 1,3-DCB was removed (Van der Meer et al. 1992). These studies show that the rate of loss of DCBs in soils and sediments is much lower when volatilization is minimized. This supports the conclusion of Wang and Jones (1994a) that biodegradation is slow compared to volatilization.

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6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to dichlorobenzenes depends in part on the reliability of supporting analytical data from environmental samples and biological specimens.

Concentrations of dichlorobenzenes in unpolluted atmospheres and in pristine surface waters are often so low as to be near the limits of current analytical methods. In reviewing data on dichlorobenzenes levels monitored or estimated in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable. The analytical methods available for monitoring dichlorobenzenes in a variety of environmental media are detailed in Chapter 7.

Due to their use and volatile nature, DCBs are detected much more frequently and at higher concentrations in air than in other environmental compartments such as soil, water, or sediment.

6.4.1 Air

1,4-DCB has been detected in indoor air, ambient outdoor air, and in occupational settings. A summary of levels of 1,4-DCB detected in indoor air is shown in Table 6-5. An update of the 1980 national ambient VOCs database prepared for the EPA summarized concentrations of 1,4-DCB by site type (Shah and Heyerdahl 1988). The median indoor air concentration of 1,4-DCB detected at 2,121 sites was 0.283 ppb (mean 3.988 ppb), and the median concentration detected from personal air monitoring of 1,650 individuals was 0.416 ppb (Shah and Heyerdahl 1988); for reference, the American Conference of Governmental Industrial Hygienists (ACGIH) Threshold Limit Value (TLV) (8-hour time-weighted average [TWA] for 1,4-DCB is 10 ppm (ACGIH 2005). The authors concluded that these values are a result of the use of 1,4-DCB in air fresheners and to control moths that could damage woolen clothing.

Because of its indoor uses, reports of indoor air monitoring show higher concentrations of 1,4-DCB than those observed in ambient outdoor air. This was also observed during the Total Exposure Assessment Methodology (TEAM) Study conducted by EPA between 1979 and 1985 in an effort to measure exposures to 20 VOCs in personal air, outdoor air, and drinking water. Data from the TEAM study were presented for the sum of 1,3- and 1,4-DCB (Wallace et al. 1986a). Because 1,4-DCB is produced and used in much greater volume than 1,3-DCB, the authors assumed that the concentrations found were almost all 1,4-DCB. The authors concluded that the major cause for the higher personal air

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Table 6-5. Levels of 1,4-Dichlorobenzene in Indoor Air

Conditions	Concentration (ppm)			Reference	
	Range	Mean	Median		
Bathroom with one deodorizer block	7.80x10 ⁻² – 1.26x10 ⁻¹			Scuderi 1986	
Bathroom with one deodorizer block in one urinal and one toilet	1.16x10 ⁻¹ – 2.20x10 ⁻¹				
Inside closet with moth flakes in closed garment bag	2.19x10 ⁻¹ – 5.45x10 ⁻¹				
Outside closet with moth flakes in closed garment bag	1.03x10 ⁻² – 7.10x10 ⁻²				
Inside wardrobe air		0.197		Morita and Ohi 1975	
Inside closet air		0.036			
Bedroom air		0.012			
2,121 Indoor sites		4x10 ⁻³	2.83x10 ⁻⁴	Shah and Heyerdahl 1988	
1,650 Personal air monitors			4.16x10 ⁻⁴		
1256 Dwellings		1.33x10 ⁻³		Brown et al. 1994	
Ventilated office air				Field et al. 1992	
Prior to pollution event	4.43x10 ⁻³ – 7.75x10 ⁻³	5.14x10 ⁻³	4.89x10 ⁻³		
During pollution event	3.54x10 ⁻³ – 7.29x10 ⁻³	4.51x10 ⁻³	4.48x10 ⁻³		
32 Smoking homes		2.79x10 ⁻³	1.51x10 ⁻⁴	5.03x10 ⁻²	Heavner et al. 1996
61 Nonsmoking homes		8.62x10 ⁻⁴	9.65x10 ⁻⁵	2.03x10 ⁻²	
757 Homes		2.61x10 ⁻³			Meek et al. 1994
12 Homes	1.66x10 ⁻⁴ – 1.78x10 ⁻²	2.50x10 ⁻³			Chan et al. 1990
Over 100 homes (United States, Germany, Netherlands)		2.16x10 ⁻³ (3.99x10 ⁻³ in the United States)		2.66x10 ⁻¹	IARC 1999
Inside four test houses			3.65x10 ⁻⁴ – 4x10 ⁻²	1.2x10 ⁻³ – 1.22x10 ⁻¹	Wallace et al. 1989
With solid deodorizer			5.64x10 ⁻²		
With spray deodorizer			6.14x10 ⁻³		
With liquid deodorizer			4.15x10 ⁻³		
With no deodorizer			4.32x10 ⁻³		
26 Normal houses		1.08x10 ⁻⁴	1.33x10 ⁻⁵	1.5x10 ⁻³	Kostiainen 1995

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Table 6-5. Levels of 1,4-Dichlorobenzene in Indoor Air

Conditions	Concentration (ppm)				Reference
	Range	Mean	Median	Maximum	
Nationwide study of Canadian homes					Fellin and Otson 1994
Winter		5.93×10^{-3}			
Spring		2.5×10^{-3}			
Summer		1.75×10^{-3}			
Fall		2.5×10^{-3}			
0 °C		3.92×10^{-3}			
0–15 °C		3.66×10^{-3}			
15 °C		2.0×10^{-3}			

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concentrations was the use of 1,4-DCB sources such as moth crystals and room deodorizers in the home (Wallace et al. 1986b).

Wallace et al. (1989) studied the influence of personal activities on exposure to VOCs. These authors reported that the median 1,4-DCB concentration in ambient outdoor air sampled 3 times/day over a 3-day monitoring period at each of three test houses was $<1 \mu\text{g}/\text{m}^3$ (0.17 ppb) and the maximum concentration was $17 \mu\text{g}/\text{m}^3$ (2.8 ppb). The median indoor 1,4-DCB air concentrations sampled individually at each of four study houses ranged from 2.2 to $240 \mu\text{g}/\text{m}^3$ (0.37–40 ppb), while the maximum concentrations ranged from 7.2 to $740 \mu\text{g}/\text{m}^3$ (1.2–123.3 ppb). The mean personal air concentration for seven individuals living in the study houses was $81 \mu\text{g}/\text{m}^3$ (13.5 ppb) (range 4.0– $240 \mu\text{g}/\text{m}^3$ [0.7–40 ppb]), while the outdoor mean 1,4-DCB personal air concentration was $1 \mu\text{g}/\text{m}^3$ (0.17 ppb). The personal air to outdoor air ratio of 81 was 4 times higher than the ratios calculated for the other VOCs tested. Two individuals living in the same house both had mean personal air concentrations of $240 \mu\text{g}/\text{m}^3$ (40 ppb); the median levels of 1,4-DCB in their breath were 40 and $47 \mu\text{g}/\text{m}^3$ (6.7 and 7.8 ppb), which was higher than the median breath level of $1.5 \mu\text{g}/\text{m}^3$ (0.3 ppb) in an individual receiving a personal exposure of $5.7 \mu\text{g}/\text{m}^3$ (1.5 ppb).

Wallace et al. (1989) further studied the activities associated with increased personal exposure to, or increased indoor air concentrations of, 1,4-DCB. The activities that increased both personal exposure and indoor air concentrations of 1,4-DCB were the use of solid toilet deodorizers, followed by spray deodorizers and liquid deodorizers, compared to the use of no deodorizers at all. The median personal exposure concentrations to 1,4-DCB were $330 \mu\text{g}/\text{m}^3$ (55 ppb) (maximum, $500 \mu\text{g}/\text{m}^3$ [83.3 ppb]), $33 \mu\text{g}/\text{m}^3$ (5.5 ppb) (maximum, $84 \mu\text{g}/\text{m}^3$ [14 ppb]), $12 \mu\text{g}/\text{m}^3$ (2 ppb) (maximum, $28 \mu\text{g}/\text{m}^3$ [4.7 ppb]), and $2.4 \mu\text{g}/\text{m}^3$ (0.4 ppb) (maximum, $6.6 \mu\text{g}/\text{m}^3$ [1.1 ppb]) for solid, spray, liquid, and no deodorizer use, respectively. Median indoor air concentrations were $340 \mu\text{g}/\text{m}^3$ (56.7 ppb) (maximum, $630 \mu\text{g}/\text{m}^3$ [105 ppb]), $37 \mu\text{g}/\text{m}^3$ (6.2 ppb) (maximum, $59 \mu\text{g}/\text{m}^3$ [9.8 ppb]), $25 \mu\text{g}/\text{m}^3$ (4.2 ppb) (maximum, $30 \mu\text{g}/\text{m}^3$ [5 ppb]), and $2.6 \mu\text{g}/\text{m}^3$ (0.43 ppb) (maximum, $5.2 \mu\text{g}/\text{m}^3$ [0.87 ppb]) for solid, spray, liquid, and no deodorizer use, respectively.

More recently, Kostianen (1995) identified >200 VOCs in the indoor air of 26 normal houses. 1,4-DCB was detected in 100% of the houses studied. 1,4-DCB was detected at a mean concentration of $0.65 \mu\text{g}/\text{m}^3$ (0.1 ppb) (median $0.08 \mu\text{g}/\text{m}^3$ [0.013 ppb], minimum $0 \mu\text{g}/\text{m}^3$ [0 ppb], and maximum $8.94 \mu\text{g}/\text{m}^3$ [1.5 ppb]) in the houses studied. Forty-eight compounds (including 1,4-DCB) were selected for further quantitative analysis in 50 normal houses and 38 “sick houses,” which had poor quality indoor air that was linked to odors and to a number of physiological follow-up study of normal and “sick houses,” the median concentration of 1,4-DCB ($0.88 \mu\text{g}/\text{m}^3$ [0.15 ppb]) in the normal houses was

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exceeded by 5–10% in 6% of the normal houses and by 10–50% in 18% of the normal houses, while in the “sick houses,” the median concentration was exceeded by 5–10% in 7.9% of the “sick houses”, by 10–50% in 2.6% of the sick houses, and by 50–200% in 5.3% of the “sick houses.” The median concentrations of 1,4-DCB reported in the 38 “sick houses” ranged from 0.00 to 5.36 $\mu\text{g}/\text{m}^3$ (0–0.89 ppb).

During a study of exposure of volatile organic compounds in the air of three photocopy centers, 1,4-DCB was detected in the breathing zone of photocopier operators at concentrations ranging from 0.1 to 3.7 ppb (Stefaniak et al. 2000). 1,4-DCB was not listed with the compounds detected in building background samples.

A nationwide study of indoor air concentrations of 26 VOC compounds was conducted in Canada in 1991 (Fellin and Otson 1994). The authors reported that mean 1,4-DCB concentrations were 35.75 $\mu\text{g}/\text{m}^3$ (5.96 ppb) (winter), 15 $\mu\text{g}/\text{m}^3$ (2.5 ppb) (spring), 10.54 $\mu\text{g}/\text{m}^3$ (1.76 ppb) (summer), and 15 $\mu\text{g}/\text{m}^3$ (2.5 ppb) (fall), and that the concentrations declined with an increase in ambient air temperature. At ≤ 0 , 0–15, and ≥ 15 °C, the 1,4-DCB mean concentrations were 23.64, 22.02, and 11.83 $\mu\text{g}/\text{m}^3$ (3.94, 3.67, and 1.97 ppb), respectively. Analysis revealed that 1,4-DCB concentrations were associated with use of household products and moth repellent crystals. These authors concluded that indoor sources of 1,4-DCB (household products and moth repellent crystal) are likely to have a more significant influence on indoor air concentrations than climatic variables. Summer conditions and outdoor temperatures >15.1 °C gave the lowest indoor air concentrations of 1,4-DCB. Moth repellent crystals are also deployed in a manner that gives reasonably constant emissions over several weeks. This compound produced a trend consistent with expected ventilation results. The highest average concentrations were observed during the winter or when temperatures were <0 °C, when ventilation is expected to be lowest. Intermediate values were measured during the fall and spring, while the lowest values were measured during the summer, when ventilation of homes is expected to be highest. Zhu et al. (2005) detected 1,2- and 1,4-DCB in the indoor air samples from 5 and 81% of 75 randomly selected dwellings in Ottawa, Canada, respectively. Arithmetic mean concentrations in these air samples were 0.77 $\mu\text{g}/\text{m}^3$ for 1,4-DCB and 0.01 $\mu\text{g}/\text{m}^3$ for 1,2-DCB.

Kinney et al. (2002) measured home outdoor, home indoor, and personal air concentrations of 1,4-DCB for selected students that attend school in the West Central Harlem section of New York City as part of the Toxic Exposure Assessment (TEACH) study. Mean winter concentrations of 1,4-DCB were 5.03 $\mu\text{g}/\text{m}^3$ in 36 home outdoor samples, 54.9 $\mu\text{g}/\text{m}^3$ in 36 home indoor samples, and 43.4 $\mu\text{g}/\text{m}^3$ in 36 personal air samples. Mean summer concentrations of 1,4-DCB were 5.03 $\mu\text{g}/\text{m}^3$ in 29 home outdoor

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samples, $54.9 \mu\text{g}/\text{m}^3$ in 36 home indoor samples, and $43.4 \mu\text{g}/\text{m}^3$ in 40 personal air samples. Similar results were obtained from TEACH study measurements in Los Angeles, California (Sax et al. 2004). Mean outdoor 1,4-DCB concentrations were $2.0 \mu\text{g}/\text{m}^3$ in 35 samples collected during the winter and $3.5 \mu\text{g}/\text{m}^3$ in 32 samples collected during the fall. Mean indoor 1,4-DCB concentrations were $40 \mu\text{g}/\text{m}^3$ in 40 samples collected during the winter and $52 \mu\text{g}/\text{m}^3$ in 32 samples during the fall. Personal air concentrations measured in Los Angeles were not reported in this study. Shendell et al. (2004) measured 1,4-DCB concentrations ranging from not detected to $3.36 \mu\text{g}/\text{m}^3$ in the air of 13 portable modular classroom structures and from not detected to $10 \mu\text{g}/\text{m}^3$ in the air of 7 main building classrooms (Shendell et al. 2004). Mean and median 1,4-DCB concentrations in air from 3 urban communities in Minnesota (Battle Creek, East St. Paul, and Phillips) were measured to be 0.1 and $0.1 \mu\text{g}/\text{m}^3$, respectively, in 132 outdoor air samples, 1.2 and $0.2 \mu\text{g}/\text{m}^3$, respectively, in 292 indoor air samples, and 3.2 and $0.4 \mu\text{g}/\text{m}^3$, respectively, in 288 personal air samples (Sexton et al. 2004).

1,4-DCB has been detected in ambient air samples in several monitoring studies, as shown in Table 6-6. Kelly et al. (1994) reported that the median concentration of 1,4-DCB was below detection limits based on 1,447 samples collected from 57 different locations. MacLeod and Mackay (1999) reported a 1,4-DCB background concentration of 3.36×10^{-5} ppm for the Southern Ontario, Canada region. The mean and median concentrations of 1,4-DCB in air from 25 sites across the state of Minnesota were 3.36×10^{-5} and 2.55×10^{-5} ppm, respectively (Pratt et al. 2000). Concentrations were not quantifiable in rural air (Shah and Heyerdahl 1988), but increasingly higher concentrations were detected in suburban and urban air. Air samples from Mexicali, Mexico, a residential industrial area, contained 1,4-DCB with concentrations ranging from 6.0×10^{-5} to 2.22×10^{-2} ppm (mean= 1.75×10^{-3} ppm), while air samples from Rosarito, Mexico, a beach resort town, contained 1,4-DCB with concentrations ranging from 2.0×10^{-5} to 1.8×10^{-4} ppm (mean= 8.0×10^{-5} ppm). Hartwell et al. (1992) reported that ambient outdoor concentrations of 1,4-DCB are considerably higher in the winter compared to the summer. The authors concluded that this effect may be due to reduced levels of sunlight in the winter, which would hinder atmospheric removal by photooxidation. Mean concentrations of 1,4-DCB in air, and in the vicinity of hazardous waste sites and sanitary landfill sites, generally average $<4.2 \times 10^{-3}$ ppm, but indoor air concentrations of 1,4-DCB may be 1–3 orders of magnitude higher where 1,4-DCB is used as a space deodorizer or moth repellent (IARC 1982; Scuderi 1986; Wallace et al. 1986a, 1986b) (see Table 6-5).

Concentrations of 1,4-DCB in workplace air were, not unexpectedly, the highest concentrations measured (IARC 1982), as shown in Table 6-7; concentrations ranged from 33–52 mg/m^3 (5.4–8.7 ppm) detected in

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Table 6-6. Levels of 1,4-Dichlorobenzene in Outdoor Air

Location	Concentration (ppm)				Reference
	Mean	Median	Maximum	Range	
Rural		0.00 ^a			Shah and Heyerdahl 1988
Semi-rural (NJ)	2.0x10 ⁻⁵ – 2.1x10 ^{-4b}		1.7x10 ⁻⁴ –4.6x10 ^{-3c}		Bozzelli and Kebbekeus 1979
Suburban		4.8x10 ⁻⁵			Shah and Heyerdahl 1988
Suburban	1.5x10 ⁻⁴			5.0x10 ⁻⁵ – 5.0x10 ⁻⁴	Delfino et al. 2003
Suburban			2.8x10 ⁻³	<1.66x10 ⁻⁴ –2.8x10 ⁻³	Wallace et al. 1989
Suburban	4.06x10 ⁻⁴				Bevan et al. 1991
Urban		5x10 ⁻⁵			Shah and Heyerdahl 1988
Urban (NJ)					Harkov et al. 1984
Summer	4x10 ⁻⁵ –7x10 ^{-5d}				
Winter	2x10 ^{-5d}				
Urban (NJ)	6x10 ^{-5d} 5x10 ⁻⁵ –6.6x10 ^{-4b}		4.3x10 ⁻⁴ –2x10 ^{-2c}		Bozzelli and Kebbekeus 1979
Urban (DC)	1.5x10 ⁻⁴		1.57x10 ⁻³		Hendler and Crow 1992
Urban	6.96x10 ⁻⁵			0.0–2.44x10 ⁻⁴	Fraser et al. 1998
Urban	1.42x10 ⁻⁴			<2.0x10 ⁻⁴ –1.3x10 ⁻³	Loscutoff and Poore 1993
Urban	0.00–7.00x10 ⁻⁵		2.20x10 ⁻⁴		Zielinska et al. 1998
Urban	2.0x10 ⁻² 2.9x10 ⁻¹		2.9x10 ⁻² 1.0x10 ¹		Grosjean 1991
Urban	4.18x10 ⁻⁴				Bevan et al. 1991
Hazardous waste sites (seven sites)	3x10 ⁻⁵ –5.4x10 ^{-4b}		4.2x10 ⁻³		Harkov et al. 1984
Hazardous waste sites and sanitary landfill sites	4x10 ⁻⁵ –5.1x10 ^{-4b} 2x10 ⁻⁵ –2.2x10 ^{-4e}		3.8x10 ⁻⁴ –4.2x10 ^{-3c}		LaRegina et al. 1986
Waste dump				1.24x10 ⁻⁵ –6.41x10 ⁻⁵	Nerin et al. 1996

^aLevel not quantifiable^bRange in arithmetic mean concentrations^cRange in maximum concentrations detected^dGeometric mean^eRange in geometric mean concentrations

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Table 6-7. Levels of 1,4-Dichlorobenzene Detected in Workplace Air

Occupation	Concentration (ppm)	
	Maximum	Range
Monochlorobenzene manufacturing plant	8.7	5.4–8.7
Abrasive-wheel plant	11.5	8–11.5
Mothball manufacturing plant	25	9–25
Chlorobenzene manufacturing plant	34	24–34
1,4-Dichlorobenzene manufacturing plant	548	12–548
Monochlorobenzene and dichlorobenzene manufacturing plant	724	–

Source: IARC 1982

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air sampled at a monochlorobenzene manufacturing facility to 4,350 mg/m³ (724 ppm) detected in air sampled at a plant manufacturing monochlorobenzene and DCB.

1,2- and 1,3-DCB have also been detected in air samples from various locations, though at much lower concentrations than 1,4-DCB. Because these isomers are not used in household products to the extent that 1,4-DCB is, they are not prevalent in indoor air. For example, mean indoor air concentrations in a ventilated office in London were approximately 3.5x10⁻³ ppm for 1,4-DCB compared to 1.4x10⁻⁴ ppm for 1,2-DCB (Field et al. 1992). Mean indoor air concentrations of 1,2-DCB from residences in some California communities were 1.39x10⁻⁵ ppm during the winter and 3.48x10⁻⁶ ppm during the summer (Pellizzari et al. 1986). 1,3-DCB was detected in the air from a university art building where there is heavy use of printmaking solvents. Mean concentrations of 1,3-DCB were 0.4 µg/m³ (median=0.8 µg/m³) on the studio floor and 0.8 µg/m³ (median below 0.5–1.5 ppb) on a non-use floor (Ryan et al. 2002). Some studies have reported 1,3-DCB air sample concentrations in combination with 1,4-DCB concentrations. However, based the production volumes of these isomers, it is expected that these concentrations represent 1,4-DCB almost entirely. The concentrations of 1,2- and 1,3-DCB measured in ambient outdoor air are shown in Tables 6-8 and 6-9, respectively. Based on the data in these tables, ambient outdoor air concentrations generally range from 0.01 to 0.1 ppb for 1,2-DCB, and from 0.001 to 0.1 ppb for 1,3-DCB. Concentrations of 1,2- and 1,3-DCB in workplace air were not located.

6.4.2 Water

DCBs have generally been detected at low concentrations in finished drinking water, surface water, and groundwater in the United States. Finished drinking water samples from 20 of the 113 cities monitored in the National Organics Monitoring Survey (NOMS) had levels of 1,4-DCB ranging from 0.01 to 1.54 ppb, with a median value of 0.03 ppb (Dressman et al. 1977), and the compound was detected in about 13% of finished drinking water supplies using surface water sources (Coniglio et al. 1980). 1,2-, 1,3- and 1,4-DCB were reported in drinking water samples from three cities on Lake Ontario at concentrations ranging from not detectable (ND) to 2 ppt, from ND to 7 ppt, and from 8 to 20 ppt, respectively (Oliver and Nicol 1982a). DCB isomers were detected in 0–3% of drinking water samples from selected locations in New Jersey, North Carolina, and North Dakota locations (Wallace et al. 1986a). Concentrations of 1,3- and 1,4-DCB were generally <1 µg/L in treated and raw water samples taken from 30 Canadian potable water treatment facilities that serve about 5.5 million consumers (Otson et al. 1982).

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Table 6-8. Levels of 1,2-Dichlorobenzene in Outdoor Air

Location	Concentration (ppm)				Reference
	Mean	Median	Maximum	Range	
Semi-rural (New Jersey)	2x10 ⁻⁵ –2.4x10 ^{-4a}		2.1x10 ⁻⁴ – 3.9x10 ^{-3b}		Bozzelli and Kebbekeus 1979
Beach resort town	3.0x10 ⁻⁵			1.0x10 ⁻⁵ – 8.0x10 ⁻⁵	Zielinska et al. 2001
Background (Southern Ontario)	1.28x10 ⁻⁶				MacLeod and Mackay 1999
25 Sites across Minnesota	1.62x10 ⁻⁵	1.28x10 ⁻⁵	2.44x10 ⁻⁵		Pratt et al. 2000
Urban (New Jersey)					Harkov et al. 1984
Summer	1x10 ⁻⁵ –3x10 ^{-5c}				
Winter	3x10 ⁻⁵ –6x10 ^{-5c}				
Urban (New Jersey)	4.8x10 ^{-5c} 2x10 ⁻⁵ –1.0x10 ^{-3a}		5.2x10 ⁻⁴ – 1x10 ^{-2b}		Bozzelli and Kebbekeus 1979
Urban (seven U.S. cities)	4.0x10 ⁻⁶ – 2.60x10 ⁻⁵			1.0x10 ⁻⁶ – 2.36x10 ⁻⁴	Singh et al. 1981a, 1981b
Urban	2.0x10 ⁻⁵			1.0x10 ⁻⁵ – 6.0x10 ⁻⁵	Zielinska et al. 2001
Urban	8.6x10 ⁻⁵			<1.0x10 ⁻⁴ – 6.0x10 ⁻⁴	Loscutoff and Poore 1993
Urban	0.0 ^d –8.80x10 ⁻⁴		1.02x10 ⁻³		Zielinska et al. 1998
Urban	1.0x10 ⁻³ – 1.3x10 ⁻¹ 5.6x10 ⁻²		1.7x10 ⁻³ – 3.1x10 ^{-1d} 6.6x10 ⁻¹		Grosjean 1991
Hazardous waste sites and sanitary landfill sites	6x10 ⁻⁵ –7.7x10 ^{-4a} 2x10 ⁻⁵ –2.3x10 ^{-4e}		6.9x10 ⁻⁴ – 8.4x10 ^{-3b}		LaRegina et al. 1986
Waste dump				1.58x10 ⁻⁵ – 9.13x10 ⁻⁵	Nerin et al. 1996

^aRange in arithmetic mean concentrations^bRange in maximum concentrations detected^cGeometric mean^dLevel not quantifiable^eRange in geometric mean concentrations

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Table 6-9. Levels of 1,3-Dichlorobenzene in Outdoor Air

Location	Concentration (ppm)				Reference
	Mean	Median	Maximum	Range	
Beach resort town			0.00 ^a		Zielinska et al. 2001
Background (Southern Ontario)	1.39x10 ⁻⁶				MacLeod and Mackay 1999
25 Sites across Minnesota	2.55x10 ⁻⁵	1.28x10 ⁻⁵	9.87x10 ⁻⁴		Pratt et al. 2000
Urban (seven U.S. cities)	4.0x10 ⁻⁶ –8.7x10 ⁻⁶			1.0x10 ⁻⁶ –4.7x10 ⁻⁵	Singh et al. 1981a, 1981b
Urban			0.00 ^a		Zielinska et al. 2001
Urban	1.01x10 ⁻⁴			<2.0x10 ⁻⁴ –3.0x10 ⁻⁴	Loscutoff and Poore 1993
Urban	0.0 ^a –8.80x10 ⁻⁴		1.02x10 ⁻³		Zielinska et al. 1998
Urban	4.0x10 ⁻³ –7.7x10 ⁻² 8.3x10 ⁻²		9x10 ⁻³ –1.5x10 ^{-1b} 2.2		Grosjean 1991
Waste dump				1.43x10 ⁻⁶ –6.70x10 ⁻⁶	Nerin et al. 1996

^aLevel not quantifiable^bRange in maximum concentrations detected

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During a national groundwater supply survey, 1,4-DCB was detected in 2 out of 280 (0.7%) random sample sites serving <10,000 persons and in 3 out of 186 (1.6%) random sample sites serving >10,000 persons above a quantitation limit of 0.5 µg/L (Westrick et al. 1984). The mean positive concentration and maximum value were 0.60 and 0.68 µg/L, respectively, for the sites serving <10,000 persons and 0.66 and 1.3 µg/L, respectively, for the sites serving >10,000 persons. 1,2- and 1,3-DCB were not detected above the quantitation limit (0.5 µg/L) in any of the random samples. 1,4-DCB was detected above 0.5 µg/L in 4 out of 321 (1.2%) nonrandom sample sites serving <10,000 persons with a median positive concentration of 0.74 µg/L and a maximum value of 0.90 µg/L. This compound was not detected above 0.5 µg/L in 158 nonrandom sample sites serving >10,000 persons. 1,2-DCB was detected above 0.5 µg/L in 1 out of 321 (0.3%) nonrandom sample sites serving <10,000 persons at a concentration of 2.2 µg/L and in 1 out of 158 (0.6%) nonrandom sample sites serving >10,000 persons at a concentration of 2.7 µg/L. 1,3-DCB was not detected above 0.5 µg/L in any of the nonrandom samples. Stackelberg et al. (2001) detected 1,2-, 1,3-, and 1,4-DCB in approximately 8, 4, and 8%, respectively, of samples collected from 30 public supply wells in southern New Jersey. Concentrations or limits of detection were not reported. 1,4-DCB had two detections at concentrations that were both below a laboratory reporting limit of 0.05 µg/L in samples from 178 active public supply wells in the Los Angeles physiographic basin (Shelton et al. 2000). 1,2- and 1,3-DCB were analyzed for, but were not detected in any of the samples from these wells. The laboratory reporting limits used for 1,2-DCB were 0.031 and 0.048 µg/L. The laboratory reporting limits used for 1,3-DCB were 0.03 and 0.054 µg/L.

1,2-DCB was detected in 0.6% of 1,077 surface water samples recorded in the STORET database at a median concentration of <10 ppb (Staples et al. 1985). 1,3-DCB was detected in 0.3% of 986 surface water samples recorded in the STORET database at a median concentration of <10 ppb. 1,4-DCB was detected in 3% of 8,576 surface water samples recorded in the STORET database at a median concentration of <0.1 ppb. 1,4-DCB was detected in 100% of 91 surface water samples from the Great Lakes at mean concentrations ranging from 0.28 ppt in Lake Huron to 1.5 ppt in Lake Ontario (IJC 1989). Oliver and Nicol (1982a) also reported concentrations of DCBs in water samples collected from the Great Lakes region. Mean 1,2-DCB concentrations were 5 ppt (range, 2–7 ppt) in samples from Lake Ontario and 6 ppt (range, ND–31 ppt) in samples from the Grand River. 1,2-DCB was not detected in samples from Lake Huron. Mean 1,3-DCB concentrations were 1 ppt (range, ND–4 ppt) in samples from the Grand River. 1,3-DCB was not detected in samples from Lake Ontario or Lake Huron. Mean 1,4-DCB concentrations were 45 ppt (range, 33–64 ppt) in samples from Lake Ontario, 4 ppt (range, 3–6 ppt) in samples from Lake Huron, and 10 ppt (range, ND–42 ppt) in samples from the Grand River. During a

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study of contaminants in 139 streams located in 30 states, 1,4-DCB was detected in 25.9% of samples in which it was searched for, with a median concentration of 0.09 $\mu\text{g/L}$ and a maximum concentration of 4.3 $\mu\text{g/L}$ (Kolpin et al. 2002).

Concentrations of 1,2-, 1,3- and 1,4-DCB from the Niagara River sampled in 1980 ranged from ND to 56 ppt, from ND to 56 ppt, and from 1 to 94 ppt. The highest concentration of 1,2- and 1,4-DCB occurred just below a chemical manufacturing plant's effluent discharge, while the highest concentration of 1,3-DCB occurred just below a waste disposal dump (Oliver and Nicol 1982a). 1,2-, 1,3-, and 1,4-DCB were also reported in waste water effluent samples collected from four plants on the Great Lakes at mean concentrations of 13 ppt (range, 6–22 ppt), 14 ppt (range, 7–13 ppt), and 660 ppt (range, 484–920 ppt) (Oliver and Nicol 1982a). In a New Jersey survey, 1,2-, 1,3- and 1,4-DCB were detected in 3, 4, and 6%, respectively, of 463 surface water samples (Page 1981). Maximum concentrations were 8.2 ppb for 1,2-DCB, 242 ppb for 1,3-DCB, and 31 ppb for 1,4-DCB. DCBs have been reported in surface waters in the vicinity of hazardous waste sites at unspecified concentrations (Elder et al. 1981) and at a concentrations of 9 ppt (1,2-DCB), 18 ppt (1,3-DCB), and 52 ppt (1,4-DCB) (Oliver and Nicol 1982a).

DCBs were monitored in wetland-treated leachate water at a municipal solid waste landfill site in central Florida from 1989 to 1990 and from 1992 to 1993 (Chen and Zoltek 1995). During the first sampling period, surface water samples contained 1,2-DCB at concentrations ranging from 0.02 to 0.10 ppb, 1,3-DCB at concentrations ranging from 0.02 to 0.10 ppb, and 1,4-DCB at concentrations ranging from 0.04 to 0.13 ppb. Groundwater samples contained 1,2-DCB at concentrations ranging from 0.09 to 1.56 ppb, 1,3-DCB at concentrations ranging from 0.08–8.95 ppb, and 1,4-DCB at concentrations ranging from 0.08 to 10.71 ppb. During the second sampling period (1992–1993), the three DCB isomers were not detected in surface water samples. 1,2- and 1,4-DCB were each detected in two groundwater samples at concentrations ranging from 0.75 to 0.84 ppb and from 0.45 to 3.74 ppb, respectively. 1,3-DCB was not detected in groundwater samples collected during the second sampling period. No detection limits were given. DCB (isomers unspecified) was detected in a study of three landfills in central Florida (Hallbourg et al. 1992). These authors reported DCB concentration ranges in groundwater of 0.37–21.2, 6–46.4, and <1–7.4 $\mu\text{g/L}$ (ppb) at three different landfill sites. Plumb (1991) reported that 1,2-, 1,3-, and 1,4-DCB were detected in groundwater collected at 36, 16, and 34 of 479 hazardous waste sites, respectively. This author reported that 1,2-DCB was detected in 240 samples collected from 36 sites in 9 of the 10 EPA regions, 1,3-DCB was detected in 82 samples collected from 16 sites in 8 of the 10 EPA regions, and 1,4-DCB was detected in 191 samples collected from 34 sites in 9 of the 10 EPA regions.

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Untreated, ambient groundwater samples from 406 urban wells and 2,542 rural wells from across the conterminous United States were collected between 1985 and 1995 as a part of the National Water-Quality Assessment Program (NAWQA) of the U.S. Geological Survey (Squillace et al. 1999). 1,2-DCB was detected in 1.4% of the urban well samples with a median concentration of approximately 0.2 µg/L (range 0.2–100 µg/L). This compound was detected in 0.2% of the rural well samples with a median concentration of approximately 1 µg/L (range 0.3–5 µg/L). 1,4-DCB was detected in 1.8% of the urban well samples with a median concentration of approximately 1 µg/L (range 0.3–50 µg/L). It was detected in 0.2% of the rural well samples with a median concentration of approximately 1.5 µg/L (range 0.6–8 µg/L). 1,3-DCB was not included in this study. Similar results were reported by Moran et al. (2004) in a summary of 1985–1999 NAWQA monitoring data involving chemical concentrations measured in 1,926 rural private wells. 1,2- and 1,3-DCB were not detected at all, while 1,4-DCB was detected in only 1 out of 1,925 samples at a concentration of 1.2 µg/L. 1,2-, 1,3-, and 1,4-DCB were detected in approximately 25, 15, and 10%, respectively, of samples collected from 95 monitoring wells in southern New Jersey, respectively (Stackelberg et al. 2001). Concentrations or limits of detection were not reported. In a separate New Jersey survey, 1,2-, 1,3-, and 1,4-DCB were detected in 3, 2, and 3 of 685 groundwater samples (Page 1981). Maximum concentrations were 6,800 ppb for 1,2-DCB, 237 ppb for 1,3-DCB, and 995 ppb for 1,4-DCB. 1,4-DCB had a frequency of detection of approximately 10% and a maximum concentration of 1.7 µg/L in groundwater samples from 29 alluvial wells beneath the Denver, Colorado area (Bruce and McMahon 1996). The authors also analyzed for 1,3-DCB, although it was not detected above the minimum detection level (0.2 µg/L) in any of the samples. 1,3-DCB was detected in two groundwater samples from five developing urban sites in the Upper Colorado River Basin with an estimated maximum concentration of 0.01 µg/L (Apodaca et al. 2002).

6.4.3 Sediment and Soil

Little information on soil concentrations of DCBs was located for the United States. One study conducted in England, however, reported DCB concentrations in agricultural soils increased during the 1960s, corresponding to a period of increased production of chlorobenzene compounds (Wang et al. 1995). The mean 1,4-DCB soil concentration reported for agricultural land was 2.17 ppb in 1942, 0.75 ppb in 1951, 1.73 ppb in 1960, 9.82 ppb in 1967, 3.9 ppb in 1972, 3.06 ppb in 1980, 1.4 ppb in 1984, and 0.4 ppb in 1991. The mean 1,3-DCB soil concentration was 0.20 ppb in 1960, 0.31 ppb in 1967, 0.36 ppb in 1972, and 0.30 ppb in 1980. 1,3-DCB soil concentrations were below the detection limit (0.2 ppb) in 1942, 1951, 1984, and 1991. 1,2-DCB soil concentrations were below the detection limit

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(0.2 ppb) during all 8 sampling years. It should be noted that 1,4-DCB has been reported to occur in soils as a result of lindane degradation (EPA 1980a; IARC 1982), so the detection of 1,4-DCB may not be indicative of 1,4-DCB disposal *per se*.

1,2-DCB was detected in 0.9% of 352 sediment samples, 1,3-DCB was detected in 0.3% of 357 sediment samples, and 1,4-DCB was detected in 2% of 357 sediment samples recorded on the STORET database (Staples et al. 1985). DCBs have been detected in sediments near hazardous waste sites (Elder et al. 1981; Hauser and Bromberg 1982). During a study of semivolatile organic compounds in streambed sediment, 1,2-DCB was detected in 0.6% of samples collected at 516 sites from 20 major river basins in the United States during 1992–1995 with a maximum concentration of 86 µg/kg (95th percentile, <50 µg/kg) (Lopes and Furlong 2001). 1,4-DCB was detected in 1.2% of samples collected at 518 sites with a maximum concentration of 140 µg/kg (95th percentile, <50 µg/kg). 1,3-DCB was not detected in samples collected from 516 sites. The concentrations of 1,2- and 1,4-DCB were both <100 µg/kg in streambed sediment samples from 9 out of 14 river sites in the New England Coastal Basin (USGS 2002). Both of these compounds were at concentrations below the minimum reporting level (50 µg/kg) in samples from the remaining five river sites. Redmond et al. (1996) detected 1,2-, 1,3-, and 1,4-DCB at concentrations up to 4.4, 7.2, and 3.6 mg/kg, respectively, in the sediment of the Calcasieu River estuary, Louisiana.

Oliver and Nicol (1982a) reported DCB concentrations in surficial sediments from 13 sites in Lake Superior, 42 sites in Lake Huron, 5 sites in Lake Erie, and 11 sites in Lake Ontario. Mean 1,2-DCB concentrations detected were 1 ppb (range, ND–1 ppb), 8 ppb (range, ND–56 ppb), 2 ppb (range, 1–4 ppb), and 11 ppb (range, 4–27 ppb) for Lakes Superior, Huron, Erie, and Ontario, respectively. Mean 1,3-DCB concentrations detected were 2 ppb (range, ND–7 ppb), 2 ppb (range, ND–14 ppb), 4 ppb (range, 1–9 ppb), and 74 ppb (range, 15–250 ppb) for Lakes Superior, Huron, Erie, and Ontario, respectively. Mean 1,4-DCB concentrations detected were 5 ppb (range, ND–9 ppb), 16 ppb (range, 2–100 ppb), 9 ppb (range, 3–20 ppb), and 94 ppb (range, 22–210 ppb) for Lakes Superior, Huron, Erie, and Ontario, respectively. These authors also reported detecting DCB concentrations in deep sediment layers in Lake Ontario from core samples from the Niagara Basin. Concentrations of 1,2-DCB in various depths of the sediment cores were as follows: 14 ppb (0–1 cm), 15 ppb (1–2 cm), 19 ppb (2–3 cm), 16 ppb (3–4 cm), 26 ppb (4–5 cm), 13 ppb (5–6 cm), and 2 ppb (6–7 cm). Concentrations of 1,3-DCB in various depths of the sediment cores were as follows: 240 ppb (0–1 cm), 330 ppb (1–2 cm), 190 ppb (2–3 cm), 48 ppb (3–4 cm), 38 ppb (4–5 cm), 17 ppb (5–6 cm), and 4 ppb (6–7 cm). Concentrations of 1,4-DCB in various depths of the sediment cores were as follows: 110 ppb (0–1 cm), 120 ppb (1–2 cm), 88 ppb (2–

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3 cm), 230 ppb (3–4 cm), 88 ppb (4–5 cm), 29 ppb (5–6 cm), and 17 ppb (6–7 cm). None of the DCBs were detected in the 7–8 cm sediment core. Chapman et al. (1996a, 1996b) also reported detecting 1,4-DCB in sediments collected around the diffuser of a large marine municipal sewage discharge outfall at Macaulay Point in Victoria, Canada. Sediment quality guidelines are set by the government to protect indigenous sediment-dwelling organisms. 1,4-DCB was detected at concentrations exceeding sediment quality guidelines (110 µg/kg [ppb] dry weight) and showed a distinctive concentration gradient, which peaked at the outfall at concentrations up to 1,710 ppb dry weight and decreased with increasing distance from the outfall. The authors attributed the source of the 1,4-DCB in the relatively untreated municipal sewage effluent to the extensive use of toilet block deodorizers.

In a recent study conducted in England, Wang and Jones (1994b) analyzed the chlorobenzene content of contemporary sewage sludge collected from 12 waste water treatment plants. Most of the plants surveyed received waste water from urban and industrial effluent and all of the sewage-treatment plants used primary treatment. 1,2- and 1,4-DCB were detected in 100% of the samples tested. 1,3-DCB was detected in 75% of the samples tested. Concentrations of 1,2-DCB ranged from 71.3 to 4,110 µg/kg (ppb) dry weight (3.57–152 ppb wet weight). For 1,2-DCB, the mean and median concentrations for the 12 plants were 877 and 237 ppb (dry weight), respectively. The authors reported that except for the monochlorobenzenes, 1,2-DCB had the highest concentration in the industrial sludges. This was believed to be the result of industrial uses of 1,2-DCB as a solvent, cleaner, degreaser, polish, and deodorant. Concentrations of 1,3-DCB ranged from below the detection limit to 467 µg/kg (ppb) dry weight (from below the detection limit to 13.5 ppb wet weight). For 1,3-DCB, the mean and median concentrations for the 12 plants were 82.3 and 30 ppb (dry weight), respectively. Concentrations of 1,4-DCB ranged from 561 to 2,320 µg/kg (ppb) dry weight (21.9–187 ppb wet weight). For 1,4-DCB, the mean and median concentrations for the 12 plants were 1,310 and 1,250 ppb (dry weight), respectively. The authors also reported that 1,4-DCB was the most abundant compound detected (exclusive of the monochlorobenzenes) and was detected at higher concentrations in the urban sludges compared to the sludges dominated by industrial sources. The authors believe that this was a result of the extensive use of the compound in moth repellent crystals, insecticides, germicides, and space deodorants. Since 1,4-DCB also has industrial uses, the absolute content of this compound was not lower in the industrial sludges as compared to the urban sludges. The authors also found that the 1,4-DCB content and that of other chlorobenzene compounds in sewage sludges from the same treatment plant were consistent over time. Wang et al. (1995) further reported that at a site in Woburn, England, sewage sludge applied to agricultural land from 1942 to 1961 contained 1,2-DCB concentrations of ND to 126 ppb (mean, 17.4 ppb; median, 6.60 ppb), 1,3-DCB concentrations of ND to 101 ppb (mean, 17.4 ppb; median, 6.60 ppb), and 1,4-DCB

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concentrations of 7.76–71.8 ppb (mean, 29.8 ppb; median, 25.5 ppb). These authors found that while concentrations of the other chlorobenzenes remained stable during the 1960s after the sludge applications were halted in 1961, the concentrations of 1,4-DCB in both the sludge-amended and control soils actually increased. The authors concluded that the 1,4-DCB could have increased in both soil plots as a result of pesticide applications since 1,4-DCB was often found as an impurity in many organochlorine pesticides or by atmospheric deposition of airborne emissions from industrial facilities or municipal waste incinerators.

6.4.4 Other Environmental Media

DCBs have been detected in meat, poultry, fish, and other types of foodstuffs. Pork meat has reportedly been tainted with a disagreeable odor and taste as a result of the use of deodorant blocks in pig stalls (EPA 1980a; IARC 1982). Eggs also have been similarly tainted after hens were exposed to 20–30 mg/m³ (3.3–5.0 ppm) of 1,4-DCB (IARC 1982). 1,4-DCB was detected in 69 out of 234 table-ready food items from the FDA's total diet study at concentrations ranging from 4.26 to 114 ppb (mean=10.7 ppb) (Heikes et al. 1995). 1,2-DCB was detected in 45 of the 234 food items at concentrations ranging from 7.80 to 24.4 ppb (mean=9.47 ppb). 1,3-DCB was detected in 6 of the food items at concentrations ranging from 5.31 to 9.76 ppb (mean=7.36). The highest level food items were chocolate chip cookies (1,4-DCB), cake doughnuts (1,2-DCB), and sandwich cookies (1,3-DCB). Page and Lacroix (1995) detected 1,4-DCB in both noncitrus based soft drinks and 10% butterfat cream at 0.1 µg/kg during a study of contaminants in Canadian foods. 1,4-DCB concentrations in different brands of butter, margarine, and peanut butter were 1.3–2.7, 12.2–14.5, and 1.2–8.8 µg/kg, respectively. Flour contained 1,2-DCB at 1.1 µg/kg and 1,4-DCB at 7.3 µg/kg, while pastry mix contained these isomers at concentrations of 1.0 and 22.0 µg/kg, respectively. Fresh food composites grown in Ontario, Canada were tested for the presence of DCBs (detection limits=0.0001 µg/g) as well as other contaminants (Davies 1988). Only 1,3-DCB was detected in fruit and root vegetables at concentrations of 0.0044 and 0.0011 µg/kg, respectively, while 1,2-DCB was the only isomer detected in the eggs/meat food group at a concentration of 0.0018 µg/kg. Both 1,3- and 1,4-DCB were detected in milk at concentrations of 0.00014 and 0.00055 µg/kg, respectively. None of the DCBs in this study were detected in leafy vegetables. The concentrations of 1,4-DCB in retail vegetables from the United Kingdom were 0.198 µg/kg (carrot cores), 0.416 µg/kg (carrot peels), 0.224 µg/kg (potato peels), 0.214 µg/kg (cauliflower stems), 0.529 µg/kg (cauliflower flowers), 0.237 µg/kg (inner lettuce leaves), and 0.118 µg/kg (outer lettuce leaves) (Wang and Jones 1994d). 1,2- and 1,3-DCB were detected only in potato cores at 0.328 and 0.096 µg/kg, respectively.

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All three DCB isomers were detected in lake and rainbow trout from the Great Lakes at concentrations ranging from 0.3 to 1 ppb for 1,2-DCB, from 0.3 to 3 ppb for 1,3-DCB, and from 1 to 4 ppb for 1,4-DCB, (Oliver and Nicol 1982a). DCBs were detected in biota collected in the vicinity of an industrial outfall in the Calcasieu River estuary, Louisiana (Pereira et al. 1988). The concentrations of 1,2-, 1,3-, and 1,4-DCB in catfish ranged from not detected to 0.11 ppm, from 0.03 to 0.19 ppm, and from 0.17 to 0.47 ppm, respectively. The concentrations of DCBs in Atlantic croakers, blue crabs, spotted sea trout, and blue catfish collected from the Calcasien River estuary were 0.08, 0.26, 0.06, and 0.06 ppm, respectively for 1,2-DCB, 0.19, 0.356, 0.09, and 0.12 ppm, respectively, for 1,3-DCB, and 0.24, 0.60, 0.90, and 2.5 ppm, respectively, for 1,4-DCB. Chung (1999) detected 1,4-DCB in the leg meat, body meat, and carapace meat of *Charybdis feriatus*, a popularly consumed edible crab in Asia, at concentrations of 0.5, 0.6, and 5.1 ppm, respectively. DCBs were detected in the edible tissue of various species of trout, nase, whiting, mullet, and pilichard fresh water fish from rivers in Slovenia and the Gulf of Triest, Yugoslavia (Jan and Movnersic 1980). 1,4-DCB concentrations in these fish ranged from trace to 0.45 ppb, while 1,2-DCB concentrations ranged from trace to 1.14 ppb. The mean upper limit of 1,4,-DCB concentrations detected in livers of flatfish (Dover sole) collected off Los Angeles, California, was <77 ppb wet weight; the mean upper limit of concentrations found in muscle tissue was <7 ppb (Young and Heesen 1978). 1,2-DCB was also detected in these fish at mean liver concentrations at or below 4.0 ppb (Young et al. 1980). Concentrations of 1,4-DCB reported in mackerel from Japanese coastal water ranged up to 0.05 ppm wet weight (50 ppb) (EPA 1980a; IARC 1982). Jori et al. (1982) reported that 1,4-DCB has been detected in carp at 0.1 ppm and in farmed fish at 0.04 ppm.

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

Inhalation is the predominant route of exposure to DCBs for the general population. According to data from the TEAM study, which includes exhaled breath measurements from about 800 individuals, 1,4-DCB was found in 44–100% of air and breath samples from several U.S. locations, and indoor air levels were up to 25 times higher than ambient outdoor levels for DCB (1,3- and 1,4-DCB) (Wallace et al. 1986b). Mean concentrations of 1,3- and 1,4-DCB measured together in breath samples collected in New Jersey and California ranged from 2.9 to 8.1 $\mu\text{g}/\text{m}^3$ (Wallace 1986b). Median concentrations of these isomers in breath samples from New Jersey, California, North Dakota, and North Carolina ranged from 0.3 to 1.3 $\mu\text{g}/\text{m}^3$ (Wallace et al. 1987, 1996). 1,2-DCB was detected above quantifiable limits (0.2–2 $\mu\text{g}/\text{m}^3$) in only 2% of the breath samples collected in New Jersey (Wallace et al. 1986c). Mean 1,2-DCB concentrations ranged from 0.08 to 0.1 $\mu\text{g}/\text{m}^3$ in breath samples collected in California (Wallace

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et al. 1988). The EPA has estimated that adult exposure to 1,4-DCB is about 35 $\mu\text{g}/\text{day}$, based on a mean ambient air concentration of 1.6 $\mu\text{g}/\text{m}^3$ (0.27 ppb) (EPA 1985a). In a separate study, average intake values for persons exposed to 1,2- and 1,3-DCB were estimated to be 1.8 and 0.8 $\mu\text{g}/\text{day}$, respectively, based on the concentrations of these substances in ambient outdoor air samples from seven large cities in the United States and a total air intake of 23 m^3/day (Singh et al. 1981a, 1981b). Inhalation exposure to 1,4-DCB may be considerably higher indoors where space deodorants or moth repellents that contain this chemical are used. Indoor inhalation exposure of the general population to 1,2- or 1,3-DCB is not expected to be important since these substances are not used in household and consumer products to the extent that 1,4-DCB is. However, one study reported that 1,3-DCB was detected in the air from a university art building where there is heavy use of printmaking solvents. Mean concentrations of 1,3-DCB were 0.4 $\mu\text{g}/\text{m}^3$ (median=0.8 $\mu\text{g}/\text{m}^3$) on the studio floor and 0.8 $\mu\text{g}/\text{m}^3$ (median below 0.5-1.5 ppb) on a non-use floor (Ryan et al. 2002). During this study, mean and median personal exposure concentrations for this compound were 2.0 and 2.3 $\mu\text{g}/\text{m}^3$, respectively.

Because water and food concentrations of DCBs are generally quite low, exposure from sources other than air is unlikely to be important. For example, drinking water containing 0.1 ppb 1,4-DCB would provide an additional intake of only 0.2 μg per day for an adult drinking 2 L of water per day. In the past, concentrations of all three DCB isomers have been detected in some freshwater fish from the Great Lakes region (Oliver and Nicol 1982a). In addition, concentrations of 1,2- and 1,4-DCB have been found in marine fishes, especially in areas near effluent discharges (Young and Heesen 1978; Young et al. 1980). However, more recent information on concentrations in edible fish and shellfish tissues is lacking.

Results of the National Human Adipose Tissue Survey (NHATS) conducted in 1982, which estimated the general population exposure to toxic organic chemicals, found that 1,4-DCB was detected in 100% of 46 composite human adipose tissue specimens analyzed at levels ranging from 12 to 500 ppb while 1,2-DCB was detected in 63% of the 46 specimens at levels ranging from <0.1–2 ppb (EPA 1986f, 1989d). These measurements indicate widespread exposure of the general population to DCBs. Using the same data, ranks for each of the 9 census regions were assigned according to the composite sample concentrations for 1,2- and 1,4-DCB or the means of multiple composite sample concentrations (Phillips and Birchard 1991). These authors reported that exposure to 1,4-DCB was highest for children (aged 0–14 years) living in the west south central (Arkansas, Louisiana, Oklahoma, and Texas), east south central (Kentucky, Tennessee, Alabama, and Mississippi), and south Atlantic regions (Delaware, Maryland, the District of Columbia, Virginia, West Virginia, North Carolina, South Carolina, Georgia, and Florida); for 15- to 44-year-olds, exposure was highest in the south Atlantic, middle Atlantic (New Jersey, New York,

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and Pennsylvania), and east north central regions (Illinois, Indiana, Michigan, Ohio, and Wisconsin); and for adults 45 years and older, exposure was highest nationally in the east south central, west south central, and east north central regions. Exposure to 1,2-DCB was highest for children (0–14 years) living in the New England (Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, and Connecticut), east north central, and west north central regions (Minnesota, Iowa, Missouri, Nebraska, Kansas, North Dakota, and South Dakota); for 15- to 44-year-olds, exposure was highest in the New England, mid Atlantic, and Pacific regions (California, Hawaii, Washington, Oregon, and Alaska); and for adults 45 years and older, exposure was highest nationally in the mid Atlantic, west north central, and west south central regions.

Table 6-10 summarizes concentrations of 1,4-DCB in blood samples from various studies. Morita and Ohi (1975) found that 1,4-DCB was present in all 34 adipose tissue and 6 blood samples taken from residents of the Tokyo, Japan metropolitan area. 1,4-DCB concentrations in the adipose tissue samples ranged from 0.2 to 11.7 ppm in the adipose tissue samples with an average concentration of 2.3 ppm and from 4 to 16 ng/ml (ppb) in the blood samples with an average concentration of 9.5 ng/mL (ppb). 1,2-DCB was detected in paired blood and biopsy fat samples obtained from 25 patients (7 male and 18 female) from British Columbia, Canada (Mes 1992). Median concentrations in whole blood, biopsy fatty tissue, blood lipids, and adipose tissue were <3.12, 28.1, <3, and 38 ppb, respectively. Maximum concentrations of 1,2-DCB in these media were 14.29, 154.5, 20,005, and 194 ppb, respectively.

Concentrations of 1,4-DCB in blood samples of 48 individuals in Alaska during February 1995 ranged from below the limit of detection (0.040 ppb) to 7.10 ppb with median values ranging from 0.02 to 0.04 ppb (Backer et al. 1997). During the Third National Health and Nutrition Evaluation Survey (NHANES III), 1,4-DCB was detected in 94.6% of 1,100 blood samples at a median concentration of 0.33 µg/L and a 95th percentile value of 9.2 µg/L (Buckley et al. 1997). Blood samples collected from July 1995 to May 1997 during the National Human Exposure Assessment Survey (NHEXAS) in EPA Region 5 (Minnesota, Wisconsin, Michigan, Illinois, Indiana, and Ohio) contained 1,4-DCB (Pellizzari et al. 2001). It was detected in approximately 80 out of 145 samples with a median concentration of 0.10 ppb, an arithmetic mean concentration of 0.38 ppb, and a maximum concentration of 45 ppb (Bonanno et al. 2001). Ashley et al. (1994, 1996) reported a mean blood level of 1,4-DCB of 1.9 ppb (median 0.33 ppb) in 1,037 samples collected from a reference group of nonoccupationally exposed individuals. Concentrations of VOCs in blood samples from a group of 126 nonsmokers and 42 smokers were also studied (Ashley et al. 1995). These authors found that mean 1,4-DCB blood levels were 3.2 ng/L (ppb) (median, 0.45 ppb; range ND–96 ppb) for nonsmokers and 2.2 ppb (median, 0.47 ppb;

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Table 6-10. Concentrations of 1,4-Dichlorobenzene in Blood Samples

Test subjects	Range (ppb)	Median (ppb)	Mean (ppb)	Reference
British Columbia, Canada (n=25)	≤14.29	<3.12		Mes 1992
Alaska, United States (n=48)	<0.040 ^a –7.10	0.02–0.04		Backer et al. 1997
NHANES III (n=1,100)		0.33		Buckley et al. 1997
EPA Region 5 (n=145)	≤45	0.10	0.38	Pellizzari et al. 2001
Non-occupationally exposed individuals (n=1,037)		0.33	1.9	Ashley et al. 1994, 1996
Nonsmokers (n=126)	ND–96	0.45	3.2	Ashley et al. 1995
Smokers (n=42)	ND–17	0.47	2.2	Ashley et al. 1995
Residents of the Love Canal area, Niagara Falls, New York	0.15–68			EPA 1985a
World Trade Center firefighters present during the collapse (n=148)			0.274	Edelman et al. 2003
World Trade Center firefighters arriving within 2 days of the collapse (n=142)			0.289	Edelman et al. 2003
World Trade Center special operations command individuals (n=95)			0.343	Edelman et al. 2003
Other World Trade Center firefighters			0.231	Edelman et al. 2003
SHIELD—children in Minneapolis, Minnesota (n=134)		0.21	4.22	Sexton et al. 2005
Adults in the United States (n=1,000)	≤49	0.33	2.1	Hill et al. 1995

^aBelow the limit of detection

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range, ND–17 ppb) for smokers. Blood levels of 1,4-DCB were not dependent on whether the subject was from the smoking or control group. All three DCB isomers have been detected in blood samples from residents of the Love Canal area in Niagara Falls, New York (IARC 1999). DCB concentrations in blood samples from nine Love Canal residents ranged from 0.15 to 68 ppb (EPA 1985a). 1,4-DCB concentrations (geometric mean) in blood samples collected from firefighters responding to the World Trade Center fire and collapse were 0.274 $\mu\text{g/L}$ for 148 firefighters who were present during the collapse and 0.289 $\mu\text{g/L}$ for 142 firefighters who arrived after the collapse (within 2 days) (Edelman et al. 2003). The mean concentrations in the blood of 95 special operations command individuals were 0.343 $\mu\text{g/L}$ compared to 0.231 $\mu\text{g/L}$ in the blood of other firefighters.

Hill et al. (1995) analyzed both blood and urine samples of 1,000 adults in the United States. These authors reported that 96% of the individuals in the study had detectable concentrations of 1,4-DCB in their blood and 98% had detectable concentrations of 2,5-dichlorophenol (the metabolite of 1,4-DCB) in their urine. 1,4-DCB levels in the blood ranged up to 49 $\mu\text{g/L}$ (ppb), with median and mean concentrations of 0.33 ppb and 2.1 ppb, respectively. Urinary 2,5-dichlorophenol concentrations ranged up to 8,700 $\mu\text{g/L}$ (ppb), with median and mean concentrations of 30 ppb and 2,000 ppb, respectively. There was a highly significant correlation ($p < 0.0001$) between 2,5-dichlorophenol in the urine and 1,4-DCB in the blood. The authors concluded that 1,4-DCB is a common, worldwide environmental contaminant. Metabolites of 1,2-DCB (2,3- and 3,4-dichlorophenol and 3,4- and 4,5-dichlorocatechol) have been detected in the urine of chemical factory workers at unspecified concentrations (Kumagai and Matsunaga 1995, 1997). These workers had been exposed to 1,2-DCB used as a solvent during the work shift prior to sample collection.

DCB (all isomers) was identified in 100% of 42 samples of human breast milk collected in five urban areas of the United States at concentrations of 0.04–68 ppb (Erickson et al. 1980). DCB (all isomers) was identified in human breast milk in 8 of 12 women who were residents of Bayonne, New Jersey (6 women), Jersey City, New Jersey (2 women), Bridgeville, Pennsylvania (2 women), and Baton Rouge, Louisiana (2 women); however, concentrations were not specified (Pellizzari et al. 1982). DCB (all isomers) was identified in breast milk samples collected from five different regions across Canada in 1982 (Mes et al. 1986). 1,2-DCB was identified in 97% of the 210 samples collected with mean and maximum milk concentrations of 3 and 29 ppb, respectively and mean and maximum concentrations in milkfat of 84 and 890 ppb, respectively. 1,3- and 1,4-DCB were identified together in 100% of the 210 samples collected with mean and maximum milk concentrations of 6 and 75 ppb, respectively and mean and maximum concentrations in milkfat of 161 and 4,180 ppb, respectively. Mean concentrations

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of 1,2-, 1,3-, and 1,4-DCB in breast milk samples collected in Slovenia, Yugoslavia in 1981 were 9, <5, and 25 µg/kg, respectively (Jan 1983). 1,2- and 1,4-DCB concentrations in the milkfat of these samples were 230 and 640 µg/kg, respectively.

Occupational exposure to DCBs may be important in several industries associated with the production of various chlorobenzene compounds. Workers may be exposed to DCBs during production, processing, and industrial use of these compounds, including the production and handling of products that contain these compounds (IARC 1999). Workplace air levels of 1,4-DCB ranging up to 4,350 mg/m³ (724 ppm) were measured at facilities producing or using the compound (IARC 1982). A summary of the levels of 1,4-DCB detected in various occupational settings is presented in Table 6-7. Currently, workers in the industries identified in Table 6-7 are likely to have the highest potential for exposure to 1,4-DCB. Levels of 1,2- and 1,3-DCB in workplace air were not found. NIOSH estimated that about 34,000 workers were potentially exposed to 1,4-DCB, about 92,000 workers were potentially exposed to 1,2-DCB, and about 400 workers were potentially exposed to 1,3-DCB in the early 1980s (NOES 1990).

6.6 EXPOSURES OF CHILDREN

This section focuses on exposures from conception to maturity at 18 years in humans. Differences from adults in susceptibility to hazardous substances are discussed in Section 3.7, Children's Susceptibility.

Children are not small adults. A child's exposure may differ from an adult's exposure in many ways. Children drink more fluids, eat more food, breathe more air per kilogram of body weight, and have a larger skin surface in proportion to their body volume. A child's diet often differs from that of adults. The developing human's source of nutrition changes with age: from placental nourishment to breast milk or formula to the diet of older children who eat more of certain types of foods than adults. A child's behavior and lifestyle also influence exposure. Children crawl on the floor, put things in their mouths, sometimes eat inappropriate things (such as dirt or paint chips), and spend more time outdoors. Children also are closer to the ground, and they do not use the judgment of adults to avoid hazards (NRC 1993).

There have been no measurements of the levels of DCBs in amniotic fluid, meconium, cord blood, or neonatal blood to investigate prenatal exposure. However, DCBs have been detected in full-term placentas collected from five regions of the Slovak Republic (Reichrtova et al. 1999, 2001). Over 40 placentas were sampled from each region. DCB concentrations measured in these placentas are provided in Table 6-11. DCBs were found most frequently in placentas from Bratislava, Slovakia

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Table 6-11. Dichlorobenzene Concentrations ($\mu\text{g}/\text{kg}$) in Human Placentas from Five Slovak Regions

Region	1,2-DCB			1,3- and 1,4-DCB		
	% Detected	Median	Maximum	% Detected	Median	Maximum
1. Bratislava	82	0.8	46.9	81	1.4	218.0
2. Nove Zamky	75	0.1	1.3	55	0.2	10.2
3. Spisska Nova Ves	10	0.0	0.2	34	0.0	45.0
4. Kosice	10	0.0	0.8	40	0.0	99.5
5. Stara Lubovna	82	8.1	64.3	79	0.8	26.9

Source: Reichrtova et al. 1999, 2001

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(industrial region—petrol, pesticide, and rubber industries), Nove Zamky, Slovakia (agricultural region with high use of fertilizers), and Stara Lubovna, Slovakia (partially agricultural rural region with increasing cross-county traffic). DCBs were found less frequently in samples from Spisska Nova Ves and Kosice (industrial regions with heavy metal pollution).

Consumption of human milk can potentially expose nursing infants to DCB. DCB (all isomers) was detected in 100% of 42 samples of human milk collected in five urban areas of the United States at concentrations ranging from 0.04–68 ppb; however, concentrations of the individual isomers were not specified (Erickson et al. 1980). DCB (all isomers) was also identified in human breast milk in 8 of 12 women who were residents of Bayonne, New Jersey (6 women); Jersey City, New Jersey (2 women); Bridgeville, Pennsylvania (2 women); and Baton Rouge, Louisiana (2 women); however, concentrations of the individual isomers were not specified (Pellizzari et al. 1982). DCB (all isomers) were identified in breast milk samples collected from five different regions across Canada in 1982 (Mes et al. 1986). 1,2-DCB was identified in 97% of the 210 samples collected with mean and maximum milk concentration of 3 and 29 ppb, respectively, and mean and maximum concentrations in milkfat of 84 and 890 ppb, respectively. 1,3- and 1,4-DCB were identified together in 100% of the 210 samples collected with mean and maximum milk concentrations of 6 and 75 ppb, respectively, and mean and maximum concentrations in milkfat of 161 and 4,180 ppb, respectively. Mean concentrations of 1,2-, 1,3-, and 1,4-DCB in breast milk samples collected in Slovenia, Yugoslavia in 1981 were 9, <5, and 25 µg/kg, respectively (Jan 1983). 1,2- and 1,4-DCB concentrations in the milkfat of these samples were 230 and 640 µg/kg, respectively.

Children are exposed to 1,4-DCB primarily by inhalation of vapors from toilet deodorants, moth proofing crystals, and moth balls used in the home or by consumption of moth balls. Consumption of DCBs in foods (see Section 6.4.4) and drinking water (see Section 6.4.2) contaminated with DCBs is thought to be a minor exposure pathway. There have been no body burden measurements made on children.

The National Human Adipose Tissue Survey (NHATS) conducted in 1982, estimated general population exposure to a variety of toxic organic chemicals. 1,4-DCB was detected in 100% of 46 composite human adipose tissue specimens analyzed at levels ranging from 12 to 500 ppb, whereas 1,2-DCB was detected in 63% of the 46 specimens at levels ranging from <0.1 to 2 ppb (EPA 1986f, 1989d). These measurements indicate widespread exposure of the general population including children (aged 0–14 years) to DCBs. Using this same data, ranks for each of the nine census regions were assigned according to the composite adipose tissue concentration of 1,4-DCB or the mean of multiple adipose composite samples

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(Phillips and Birchard 1991). These authors reported that exposure to 1,4-DCB based on adipose tissue levels was highest nationally for children (aged 0–14 years) in the west south central (Arkansas, Louisiana, Oklahoma, and Texas), east south central (Kentucky, Tennessee, Alabama, and Mississippi), and south Atlantic regions (Delaware, Maryland, the District of Columbia, Virginia, West Virginia, North Carolina, South Carolina, Georgia and Florida) as compared to other areas of the United States. Exposure to 1,2-DCB was highest for children (0–14 years) living in the New England (Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, and Connecticut), east north central (Illinois, Indiana, Michigan, Ohio, and Wisconsin), and west north central regions (Minnesota, Iowa, Missouri, Nebraska, Kansas, North Dakota, and South Dakota). 2,5-Dichlorophenol, a metabolite of 1,4-DCB, and 3,4-dichlorophenol, a metabolite of 1,2-dichlorophenol, were detected in urine samples from 197 Arkansas children (Hill et al. 1989). 2,5-Dichlorophenol was detectable in 96% of the samples with median and maximum concentrations of 9 and 1,200 ppb, respectively. 3,4-Dichlorophenol was detectable in 6% of the samples with median and maximum concentrations of <1 ppb (detection limit) and 9 ppb.

Childhood exposures can be reduced by appropriate use of 1,4-DCB-containing compounds in the home and appropriate supervision of young children. Small children, because of their hand-to-mouth activity, may receive significant exposure from ingestion of 1,4-DCB. Moth balls look like candy; a young child may be tempted to eat them. Accidental poisoning by consumption of this household chemical is likely to occur if the moth balls and/or crystals are placed in a location easily accessed by children and under conditions where children are not properly supervised. It is also important that children not be allowed to play around toilet deodorants and air fresheners unsupervised. Since some 1,4-DCB is applied as a crystalline form, children may be exposed dermally, orally (in hand-to-mouth activities), or by inhalation of dust particles or vapors while playing on floors or carpeting where 1,4-DCB-contaminated particles may have fallen after moth proofing activities in the home. It is important that children not be allowed entry into 1,4-DCB-treated storage areas until the moth crystals have sublimated and the vapors have dissipated.

Children living in homes of adults that are occupationally exposed to DCBs must not be exposed to the contaminated work clothes or shoes of adults (DHHS 1995). While the vast majority of occupational exposures are likely to be by inhalation of DCB vapors by workers, a potential route of exposure to other members of the worker's family including children may occur if DCB contaminated work clothes are brought home for laundering. The chemical contamination on the clothing may then vaporize releasing DCBs into the indoor air of the workers' home. Occupational protection statements for the end use DCB products state that individuals occupationally exposed to these products should take off all wet or

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contaminated work clothes and shoes and shower using soap and water, and then put on clean clothes (NIOSH 1997). Although no studies were found that investigated this pathway of exposure, it is conceivable that poor hygiene practices among occupationally exposed adults could potentially result in domestic exposures of other family members to DCBs carried home on work clothes and subsequently to the vapors released.

As discussed in Section 6.5 of this profile, inhalation of indoor air is the major exposure route for both adults and children in the general population; however, several other minor pathways may also result in exposure. Like adults, children living in proximity to hazardous waste sites may be exposed to DCBs in contaminated groundwater. If residential wells are the primary source of drinking water, this may pose a risk to human health by consumption of contaminated water and by increased inhalation of, and dermal contact with DCBs during showering and bathing.

Little information on the levels of DCB concentrations in infant and toddler foods and in baby formula was located. Page and Lacroix (1995) analyzed a variety of beverage and food samples for 32 different volatile contaminants, including 1,4-DCB, and found residue levels to be quite low (range, 0.1–22 ppb). Soft drink samples contained 0.1 µg/kg (ppb), while cream with 10% butterfat, butter, margarine, peanut butter, flour, and pastry mix contained concentrations of 0.1, 1.3–2.7, 12.2–14.5, 1.2–8.8, 7.3, and 22 ppb, respectively. 1,2-, 1,3-, and 1,4-DCB were detected in 45, 6, and 69 out of 234 table-ready food items from the FDA's total diet study, respectively. Positive detections of all three isomers had concentrations within a range of 4.26 to 114 ppb (Heikes et al. 1995). No information was located to determine whether children differed in their weight-adjusted intake of 1,4-DCB.

There are some parental exposures to DCBs that might result in potential exposures of children to this chemical. DCBs are not genotoxic and, thus, there should be no concern about exposure to parental germ cells (see Table 3-3 and 3-4 for further information). Additional information on the genotoxicity of these compounds can be found in Section 3.7, Children's Susceptibility. Because DCBs have been widely detected in samples of human adipose tissue, the potential exists for these compounds to be stored in maternal tissues from preconception exposures and mobilized during gestation or lactation so that the developing fetus or embryo or nursing infant is exposed even after external exposure to the mother has ceased. Like all organochlorine compounds, DCBs are stored in fatty tissue. 1,4-DCB was detected in 100% of adipose tissue samples of adults and children analyzed as part of the National Adipose Tissue study (EPA 1986f). As previously mentioned, there have been measurements of all DCB isomers

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(combined) in human breast milk (Erickson et al. 1980; Pellizzari et al. 1982). For additional information on developmental effects of this compound, please see Section 3.7, Children's Susceptibility.

During the Minnesota Children's Pesticide Exposure Study, 1,4-DCB was detected above $0.2 \mu\text{g}/\text{m}^3$ in 70 of 73 personal air samples, 83 of 101 indoor air samples, and 42 of 100 outdoor air samples collected from households with children (Adgate et al. 2004). The mean concentration of 1,4-DCB was $1.4 \mu\text{g}/\text{m}^3$ in the personal air samples, $0.9 \mu\text{g}/\text{m}^3$ in the indoor air samples, and $0.3 \mu\text{g}/\text{m}^3$ in the outdoor air samples. During the School Health Initiative: Environment, Learning, Disease (SHIELD) study, the median concentrations of 1,4-DCB measured in the outdoor home air, indoor school air, indoor home air, and personal air of 113 children from two inner-city schools in Minneapolis, Minnesota were 0.1, 0.5, 0.7, and $1.0 \mu\text{g}/\text{m}^3$, respectively, during the winter and 0.2, 0.5, 0.9, and $1.3 \mu\text{g}/\text{m}^3$, respectively, during the summer (Adgate et al. 2004). The mean, median, and 95th percentile concentrations of 1,4-DCB measured in the blood of 134 children during the SHIELD study were 4.22, 0.21, and $24.5 \mu\text{g}/\text{m}^3$, respectively (Sexton et al. 2005).

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

In addition to individuals who are occupationally exposed to DCBs (see Section 6.5), several groups within the general population have potentially higher exposures (higher than background levels) to DCBs than the general population. These populations include individuals living near sites where DCB are produced or used in manufacturing and sites where DCBs are disposed.

Those individuals living or working near industrial facilities or hazardous waste sites with higher than average levels of DCBs in the air would have the potential for above-average exposures. In addition, individuals using space deodorants (air fresheners), toilet block deodorants, or moth repellents (moth balls or crystal) containing 1,4-DCB in their homes have the potential for high exposure to this compound (Scuderi 1986). Indoor air concentrations resulting from the use of these products in bathrooms and closets have been measured at levels up to $1.3 \text{ mg}/\text{m}^3$ (0.22 ppm) (Scuderi 1986).

Individuals living in proximity to hazardous waste sites may also be exposed to DCB by contaminated groundwater. If residential wells are the primary source of drinking water, this may pose a risk to human health by consumption of contaminated water and by increased inhalation of and dermal contact with DCBs during showering and bathing.

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6.8 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA, as amended, directs the Administrator of ATSDR (in consultation with the Administrator of EPA and agencies and programs of the Public Health Service) to assess whether adequate information on the health effects of dichlorobenzenes is available. Where adequate information is not available, ATSDR, in conjunction with NTP, is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of dichlorobenzenes.

The following categories of possible data needs have been identified by a joint team of scientists from ATSDR, NTP, and EPA. They are defined as substance-specific informational needs that if met would reduce the uncertainties of human health assessment. This definition should not be interpreted to mean that all data needs discussed in this section must be filled. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

6.8.1 Identification of Data Needs

Physical and Chemical Properties. The physical and chemical properties of the DCBs are sufficiently well characterized to allow estimation of its environmental fate (Amoore and Hautala 1983; Chiou et al. 1983; Howard 1989; Lide and Frederikse 1994; Newsom 1985; NFPA 1994; Sax and Lewis 1987; Schwartzenbach and Westall 1981; Verschueren 1983; Wilson et al. 1981). On this basis, it does not appear that further research in this area is required.

Production, Import/Export, Use, Release, and Disposal. Data on the production and uses of DCBs in the United States are available (CMR 1990; HSDB 2005; IRPTC 1985; SRI 1996; TRI03 2005). Incineration is the recommended disposal method for DCBs (HSDB 2005; IRPTC 1985). Disposal of this compound is controlled by federal regulations (HSDB 2005; IRPTC 1985). Available information appears to be sufficient for assessing the potential for release of, and exposure to, DCBs.

According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit substance release and off-site transfer information to the EPA. The TRI, which contains this information for 2003, became available in May of 2005. This database is updated yearly and should provide a list of industrial production facilities and emissions.

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Environmental Fate. The environmental fate of the DCBs has been well characterized. Their volatilization into air from other media, reaction with hydroxyl radicals in the atmosphere, transport through soil, and biodegradation by water and soil microorganisms seem to be well understood (Bouwer and McCarty 1982, 1983, 1984; Chiou et al. 1983; Cuppitt 1980; EPA 1985d; Garrison and Hill 1972; Howard 1989; Ligocki et al. 1985; Newsom 1985; Schwartzbach and Westall 1981; Singh et al. 1981a, 1981b; Scuderi 1986; Spain and Nishino 1987; Tabak et al. 1981; Wakeham et al. 1983; Wang and Jones 1994a, 1994b, 1994c; Wilson et al. 1981). Volatilization, sorption, biodegradation, and bioaccumulation appear to be competing processes for the removal of DCBs from water (Spain and Nishino 1987). Additional data on the rates of these reactions under various environmental conditions would be useful, but do not appear to be essential to understand the behavior of DCBs in the environment.

Bioavailability from Environmental Media. DCBs have been shown to be well absorbed by laboratory animals via inhalation and oral exposure (Hawkins et al. 1980; Kimura et al. 1979). No information has been located regarding absorption by the dermal route. Although no information has been located on the absorption of this substance from breathing contaminated air or ingesting DCBs that are contained in soil or plant material are expected to be well absorbed from these media. It would be useful to have information on whether, and to what extent, absorption of DCBs can occur as a result of dermal contact with soil or from swimming in surface water or bathing or showering in groundwater that contains DCBs.

Food Chain Bioaccumulation. Bioconcentration of DCBs has been documented for several aquatic species (ASTER 1995; Chiou 1985; Oliver and Nicol 1982a; Oliver and Niimi 1983). Based on the relatively high K_{ow} , it appears that bioaccumulation does occur (Leo et al. 1971). Oliver and Nicol (1982a) measured concentrations of chlorobenzenes in sediments, water, and selected fish from the Great Lakes. Their limited fish analyses indicate that chlorobenzenes, including DCBs, are bioconcentrated by fish, but to a much smaller extent than compounds such as DDT or PCBs. DCBs have also been shown to be accumulated by terrestrial plants (Wang et al. 1996). No data were located on biomagnification of DCBs through terrestrial or aquatic food chains. Additional information on bioconcentration of DCBs by commercially important fish, shellfish, and plant species and biomagnification would be helpful in evaluating the potential importance of food chain bioaccumulation to human exposure.

Exposure Levels in Environmental Media. Several studies are available documenting levels of DCBs in indoor and ambient outdoor air, water, and soil and sediments in rural, suburban, and urban areas and in the environs of hazardous waste sites (Bozzelli and Kebbekus 1979; Coniglio et al. 1980;

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Dressman et al. 1977; Elder et al. 1981; Fellin and Otson 1994; Harkov et al. 1984, 1985; Hauser and Bromberg 1982; IARC 1982; IJC 1989; Kostianen 1995; LaRegina et al. 1986; Oliver and Nicol 1982a; Page 1981; Scuderi 1986; Shah and Heyerdahl 1988; Staples et al. 1985; Wallace et al. 1986a, 1986b, 1989). It would be valuable to have more recent monitoring data to better estimate the potential for current human exposure levels from these media, especially in the vicinity of hazardous waste sites.

Although there is little information on DCB levels in food (IARC 1982; Oliver and Niimi 1983; Page and Lacroix 1995), it does not appear that this is an important source of human exposure. However, additional data on DCB levels in foodstuffs, especially commercially important fish, shellfish, and plants, would be useful to confirm this assumption.

Reliable monitoring data for the levels of dichlorobenzenes in contaminated media at hazardous waste sites are needed so that the information obtained on levels of dichlorobenzenes in the environment can be used in combination with the known body burden of dichlorobenzenes to assess the potential risk of adverse health effects in populations living in the vicinity of hazardous waste sites.

Exposure Levels in Humans. Detection of DCBs in breath, adipose tissue, breast milk, and blood can be used as indicators of human exposure (Ashley et al. 1994, 1995; EPA 1986f, 1989d; Erickson et al. 1980; Hill et al. 1995; Pellizzari et al. 1982; Wallace et al. 1986b). Levels of DCBs in breath appear to provide rough estimates of recent preceding exposure (Wallace et al. 1986b), while levels in adipose tissue may be useful to indicate less recent past exposure (EPA 1986f, 1989d). The level of 2,5-dichlorophenol (a metabolite of 1,4-DCB) has also been reported in urine of 1,000 individuals (Hill et al. 1995), and is highly correlated to 1,4-DCB in blood. Additional data correlating levels in environmental media with human tissue levels, particularly for populations living in the vicinity of hazardous waste sites that contain DCBs, would be helpful in establishing levels of the chemical to which humans have been exposed. Additional monitoring data on the occupational exposure of workers to DCBs would be helpful. Additional studies reporting inhalation exposure through the use of toilet air fresheners and mothballs that contain DCBs would be useful.

This information is necessary for assessing the need to conduct health studies on these populations.

Exposures of Children. Children, like all members of the general population, are exposed to DCBs primarily by inhalation. No exposure or body burden studies were specifically located related to children. Studies to quantify the amount of DCBs in amniotic fluid, meconium, cord blood, or neonatal blood

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would be useful in assessing prenatal exposure. Maternal-fetal exposure should be evaluated since there is some genotoxic potential. Studies on the amount of the DCBs specifically in breast milk would be useful in assessing exposures in nursing infants. Although inhalation of 1,4-DCB is the most important exposure pathway in humans, consumption of moth crystals or moth balls by young children also may result in additional exposure of concern. It is not known whether children are different from adults in their weight-adjusted intake of 1,4-DCB. Studies on this topic with respect to inhalation and dietary intake are needed. Childhood exposure to this chemical can be decreased by the appropriate use of this compound particularly in the home and by appropriate supervision of young children. Education programs for parents and young children may be appropriate to reduce poisoning incidents. Studies on exposures of janitorial personnel and other occupationally exposed adults would also be helpful in determining the amount of 1,4-DCB that may accumulate on work clothes and whether crystalline particles of the toilet deodorants or moth crystal can be carried home on work clothing leading to additional domestic exposures from crystals and subsequently to vapors.

Child health data needs relating to susceptibility are discussed in Section 3.12.2, Identification of Data Needs: Children's Susceptibility.

Exposure Registries. No exposure registries for dichlorobenzenes were located. This substance is not currently one of the compounds for which a sub-registry has been established in the National Exposure Registry. The substance will be considered in the future when chemical selection is made for sub-registries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to exposure to this substance.

6.8.2 Ongoing Studies

A search of Federal Research in Progress (FEDRIP 2005) identified one ongoing study that is related to dichlorobenzenes. James Heist of Ftc Acquisition Corporation is being funded by the Air Force to study material recycling and waste minimization using a freeze crystallization process. Dichlorobenzenes are among the substances for which recycling via this method will be considered.