5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Chloroform is both a synthetic and naturally occurring compound, although anthropogenic sources are responsible for most of the chloroform in the environment. Chloroform is released into the environment as a result of its manufacture and use; its formation in the chlorination of drinking water, municipal and industrial waste water, and swimming pool and spa water; and from other water treatment processes involving chlorination. Under anaerobic conditions, some bacteria can dehalogenate carbon tetrachloride to release chloroform. Most of the chloroform released into the environment will eventually enter the atmosphere. In the atmosphere, chloroform may be transported long distances before ultimately being degraded by indirect photochemical reactions with such free radicals as hydroxyl. The compound has been detected in ambient air in locations that are remote from anthropogenic sources. Chemical hydrolysis is not a significant removal process. While microbial biodegradation can take place, such reactions are generally possible only at fairly low concentration levels due to chloroform's toxicity. Microbial biodegradation of chloroform may also be inhibited due to high levels of other aromatics (e.g., toluene), chlorinated hydrocarbons (e.g., trichloroethylene [TCE]), or heavy metals (e.g., zinc). Because of its low soil adsorption and slight, but significant, water solubility, chloroform will readily leach from soil into groundwater. In groundwater, chloroform is expected to persist for a long time.

The general population is exposed to chloroform by ingesting water and food, inhaling contaminated air, and possibly through dermal contact with chloroform-containing water. Generalizations can be made concerning the chloroform concentrations in the environment. Background air concentrations appear to be in the sub-ppb range, but certain urban, indoor, and source-dominated areas may show elevated concentrations when compared to background concentrations. Drinking water levels as high as 311 ppb have been reported in public water supplies, although most of the reported concentrations are less than 50 ppb, typically ranging between 2 and 44 ppb. Levels in drinking water derived from groundwater contaminated with leachate from landfills and hazardous waste sites can sometimes be much higher. Except for a few special surveys, regular testing for chloroform or other trihalomethanes (THMs) has focused on larger community water treatment systems serving at least 10,000 people. Very limited information was located regarding the concentrations found in ambient soil. Chloroform has also been detected in the ppb range in certain foods.

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Occupational exposure to higher than background levels of chloroform can be expected to occur in some occupations although few quantitative exposure data were located. Populations with the highest potential' exposures appear to be workers employed in or persons living near industries and facilities that manufacture or use chloroform; operators and individuals who live near municipal and industrial waste water treatment plants and incinerators, and paper and pulp plants; and persons who derive their drinking water from groundwater sources contaminated with leachate from hazardous waste sites.

Chloroform has been identified in at least 717 of 1,428 current or past EPA National Priorities List (NPL) sites (HazDat 1996). However, the number of sites evaluated for chloroform is not known. Figure 5-l shows the distribution of sites in the continental United States; there are 710 such sites. In addition, there are 6 sites in the Commonwealth of Puerto Rico and one site in the Virgin Islands (not shown in Figure 5-l).
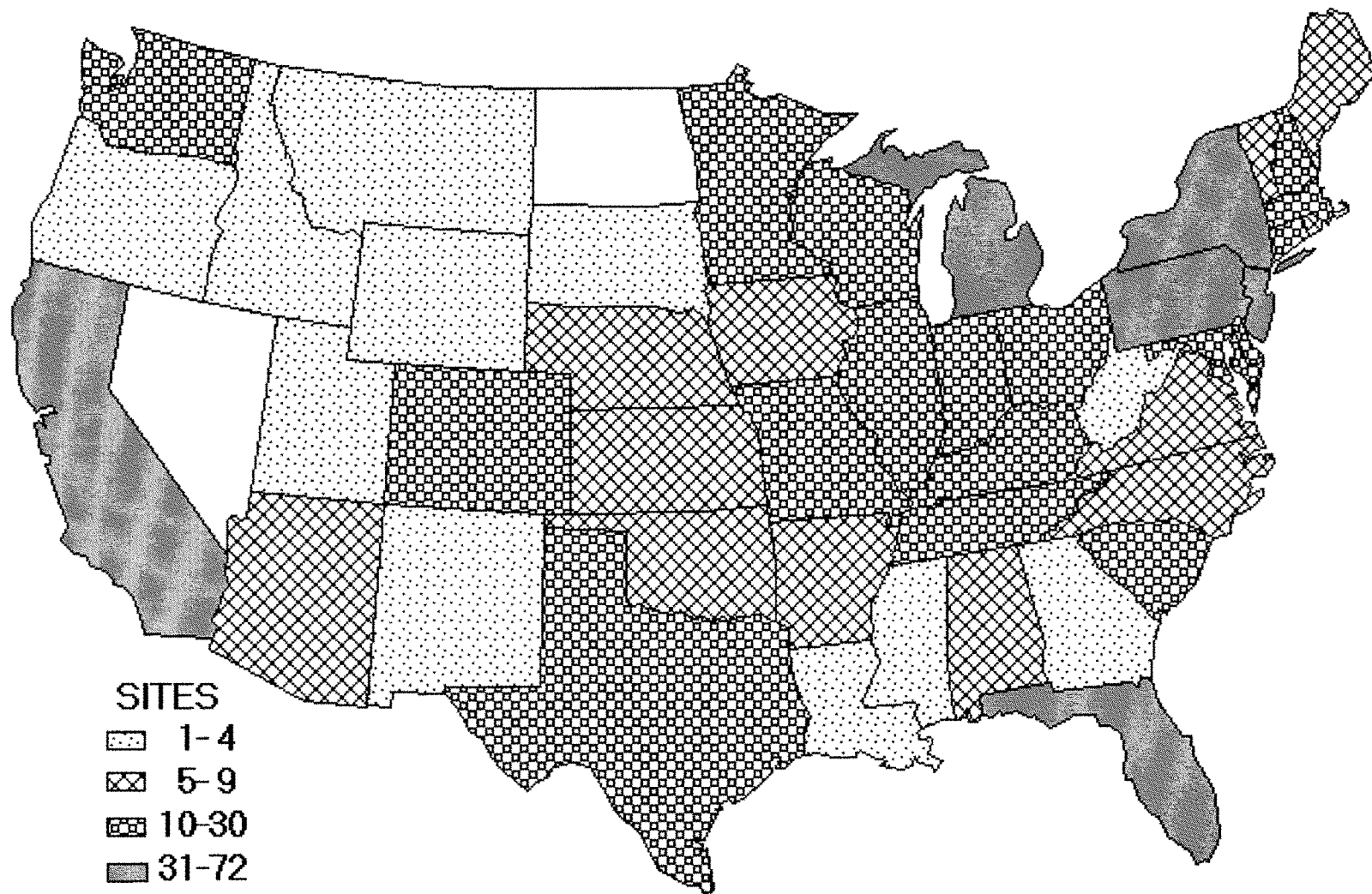
5.2 RELEASES TO THE ENVIRONMENT

5.2.1 Air

According to the 1991 Toxic Chemical Release Inventory (TR191 1993), releases of chloroform to the air from 182 large processing facilities were 17,034,926 pounds (8,413,971 kg). This represented about 95% of the total releases to environmental media. For TRI information from 1993 (TR193 1995), 13,485,992 pounds (6,068,696.4 kg) were reported as released to the air from 172 facilities, which represents around 97.1% of the total releases to environmental media. Since there was a slight decrease in overall chloroform production in 1993 as compared with 1991 (CEN 1995), it is difficult to attribute this decrease in releases to the air to improved pollution prevention measures as opposed to short-term fluctuations related to production capacity utilization factors. The releases of chloroform to air from facilities that manufactured and processed it in the United States during 1993 are reported in Table 5-l (TR193 1995). The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list.

Current comprehensive quantitative data or estimates of chloroform releases to the atmosphere are lacking. Some direct releases to the atmosphere are expected to occur during the manufacture, loading, and transport of chloroform (EPA 1985a, 1985b). Indirect chloroform releases have resulted from its use in the manufacture of hydrochlorofluorocarbon-22, fluoropolymers, pharmaceuticals,

Figure 5-1. Frequency of NPL Sites with Chloroform Contamination

SITES
- 1-4
- 5-9
- 10-30
- 31-72

Derived from HazDat 1996

Table 5-1. Releases to the Environment from Facilities That Manufacture or Process Chloroform

		Range of reported amounts released in pounds per year [a]						
State [b]	Number of facilities	Air	Water	Land	Underground injection	Total environment [c]	POTW transfer	Off-site waste transfer
AK	2	140697-240000	19000-40869	0	0	181566-259000	0	0
AL	12	250-620000	0-16000	0-2000	0	250-636000	0	0-132346
AR	6	1-229000	0-3600	0-5	0	1-229960	0	0-250
AZ	1	23000	0	22400	0	45400	0	0
CA	2	0-3150	0-3600	0	0	0-6750	0	0
CO	1	500	0	0	0	500	250	67000
CT	1	2250	14400	0	0	16650	0	242520
FL	7	62-235000	0-11100	0-3700	0	62-239120	0-53580	0-18500
GA	6	21199-329000	0-4500	0-750	0	21949-333273	0	0-130
ID	1	88000	750	5	0	88755	0	0
IL	2	108-2150	0	0	0	108-2150	0-500	12242-160130
IN	3	250-9700	0-30	0	0	250-9730	0	5-99000
KS	2	40000-44662	0	0	0-37950	40000-82612	0	781-7726
KY	7	0-150000	0-7200	0-5	0	0-157205	0-849	0-89630
LA	16	3-280000	0-16600	0-250	0-15	34-284950	0	0-72280
MA	1	17650	0	0	0	17650	0	975
MD	1	17000	0	0	0	17000	32000	0
ME	7	28933-441000	650-2200	0-130	0	30377-441756	0	0-370
MI	5	2475-55000	2-570	0-79	0	2500-55600	0-15000	0-596
MN	3	8400-140000	0-4000	0-7	0	12407-140000	0-18417	0-1600
MO	2	8750-105000	0	0	0	8750-105000	250-3400	32100-114500
MS	2	25510-292000	1269-15000	1-5	0	26780-307005	0	0
MT	1	49800	700	0	0	50500	0	0
NC	4	98000-730000	0-3200	0	0	98000-730650	0-5	0-730
NH	1	36953	79	18	0	37050	0	0
NJ	4	150-18200	0-439	0	0	150-18639	0-13350	0-44300
NY	3	10-11000	0-250	0	0	10-11250	0-12	250-28729
OH	3	406-22100	0-820	0	0	406-22920	0	0-4037

Table 5-1. Releases to the Environment from Facilities That Manufacture or Process Chloroform (continued)

Range of reported amounts released in pounds per year [a]

State[b]	Number of facilities	Air	Water	Land	Underground injection	Total environment[c]	POTW transfer	Off-site waste transfer
OK	1	93200	700	550	0	94450	0	0
OR	3	1000-143800	0-3000	0	0	1000-146800	0-100000	250-16000
PA	7	10700-231000	0-2900	0-15	0	10700-232705	0-70000	0-55700
PR	6	0-92072	0	0	0	0-92072	0-161519	0-306374
SC	5	500-136540	0-5649	0	0	500-138440	0-5	0-500
TN	2	38000-47763	1900-7441	0-160	0	39900-55364	0	0-3
TX	17	5-600000	0-5350	0-900	0-74	5-600000	0-112000	0-377935
VA	4	55950-187000	31-2600	0-170	0	57979-187063	0	0
VT	1	4	0	0	0	4	0	168700
WA	7	5036-196000	0-15000	0-5	0	12275-200500	0	0
WI	11	1000-150000	0-4900	0-250	0	1000-152500	0-13	0-51000
WV	3	870-22276	0-654	0-200	0	870-23130	0-1	0-1328

Source: TRI93 1995

[a] Data in TRI are maximum amounts released by each facility.
[b] Post office state abbreviations used
[c] The sum of all releases of the chemical to air, land, water, and underground injection wells by a given facility

POTW = publicly owned treatment works

ethylene dichloride, dyes, and fumigants (Deshon 1979; EPA 1985a, 1985b; HSDB 1996). Chloroform releases result from its formation and subsequent volatilization from chlorinated waters including drinking water, municipal and industrial waste waters, process waters and effluent from the bleaching of pulp in pulp and paper mills, cooling-tower water, and swimming-pool and whirlpool-spa water (Benoit and Jackson 1987; EPA 1985a, 1985b; Hoigne and Bader 1988). Increased release rates of the chloroform in waters can be expected from chloroform-containing waters that are heated (e.g., water used for cooking, showers, swimming pools, and spas). Aeration and use of groundwater contaminated with chloroform are a potential source of emission to the atmosphere (Crume et al. 1990). Chloroform is released as a result of hazardous and municipal waste treatment processes. The chloroform released may have initially been present in the waste or possibly formed during chlorination treatment (Corsi et al. 1987; EPA 1990b; Namkung and Rittmann 1987). Releases may also occur from hazardous waste sites and sanitary landfills where chloroform was disposed, and from municipal and hazardous waste incinerators that bum chloroform-containing wastes or produce chloroform during the combustion process (LaRegina et al. 1986; Travis et al. 1986).

In the past, minor releases may have resulted from the use of consumer products (e.g., certain air deodorizers and cleaning products) that contained chloroform as a component or residual product (Bayer et al. 1988; Wallace et al. 1987a). Chloroform is widely used in laboratory work as an extractant. It is also still used in certain medical procedures, such as dental root canal surgeries (McDonald and Vire 1992), and in combination with aspirin as an experimental treatment for serious cases of herpes zoster (King 1993). These medical uses are extremely limited and would contribute very minor amounts of chloroform as releases to the air.

5.2.2 Water

In 1991, releases of chloroform to the water from as many as 167 large processing facilities were 654,452 pounds (323,250 kg) (TR191 1993), amounting to about 3.6% of total releases to all environmental media. TRI information from 1993 (TR193 1995) indicates releases of chloroform to water from as many as 165 facilities were 335,032 pounds (150,764.4 kg) (TR193 1995), or about 2.4% of the total releases to all environmental media. The releases of chloroform to water from facilities that manufactured and processed it in the United States during 1993 are reported in Table 5-l (TR193 1995). The TRI data indicate that only a small fraction of the chloroform released to the

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environment is released to water. The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list.

Current, more comprehensive quantitative data or estimates of chloroform releases to natural waters are lacking. Direct releases to water are expected via waste waters generated during chloroform manufacture and its use in the manufacture of other chemicals and materials (EPA 1985a). Direct discharge sources are expected to be relatively minor contributors to total chloroform emissions to water relative to the formation of chloroform resulting from the chlorination of drinking water or chlorination to eliminate pathogens in discharged wastes or other process waters (EPA 1985a). Since chlorination to disinfect water supplies is nearly universal, chloroform contamination resulting from chlorination will also be widespread (see discussion on levels monitored or estimated in water in Section 5.4.2).

Other chloroform emission sources tend to be relatively isolated point sources. Chlorination of municipal and industrial waste waters at municipal and industrial waste water-treatment plants, process waters and effluent from the bleaching of pulp in pulp and paper mills, cooling-tower water, and swimming-pool and whirlpool-spa water will also result in chloroform formation (Benoit and Jackson 1987; Comba et al. 1994; EPA 1985a, 1985b, 1990a; Hoigne and Bader 1988). The use of modern treatment facilities may reduce the amounts of chloroform released to environmental waters. This has been demonstrated at a modern kraft pulp mill (Paasivirta et al. 1988); however, much of the chloroform removed from the waste water may be released to the atmosphere by volatilization. Release of chloroform to groundwater has resulted from improper disposal of chloroform-containing waste at hazardous waste sites (Clark et al. 1982; Dewalle and Chian 1981; Harris et al. 1984; Sawhney 1989). An additional minor source of water contamination may be atmospheric rainout since chloroform has been found in rainwater (Kawamura and Kaplan 1983). Other sources of chloroform release to surface water include breweries, thermal combustion of plastics, reaction of dissolved chlorine with sediment and other materials in water, biological production by marine algae, and the reaction of chlorinated pollutants with humic materials in natural waters (EPA 1985a).

5.2.3 Soil

In 1991, releases of chloroform to the land from as many as 137 large processing facilities were 28,582 pounds (14,117 kg), which amounts to less than 1% of the total releases to all environmental

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media (TR191 1993). TRI information from 1993 (TR193 1995) indicates releases to the land from as many as 124 facilities was 32,926 pounds (14,816.7 kg), which is less than 1% of the total releases to all environmental media. The releases of chloroform to soil from facilities that manufactured and processed it in the United States during 1993 are reported in Table 5-l (TR193 1995). The TRI data indicate that only a very small fraction of the chloroform released to the environment is released to land. The TRI data should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list.

Current comprehensive quantitative data or estimates of chloroform releases to soil are lacking. Chloroform releases to soil have occurred at hazardous waste sites containing improperly disposed wastes where chloroform has leached through soil to groundwater (Clark et al. 1982; Dewalle and Chian 1981; Harris et al. 1984; Sawhney 1989). Land disposal of sludge from municipal and industrial waste water-treatment plants may also result in chloroform releases to soil (EPA 1990a). Direct land disposal of chloroform-containing wastes may have occurred in the past, but land disposal of chloroform wastes is currently subject to restrictive regulations (EPA 1988a, 1989b). An additional minor source of soil contamination may be atmospheric rainout since chloroform has been found in rainwater (Kawamura and Kaplan 1983).

Chloroform has been used as a carrier or solvent for some pesticides (HSDB 1996). It is still used as a carrier for at least one pesticide formulation with dichlorvos as the active ingredient (Petrelli et al. 1993). Application of pesticides using chloroform in the carriers could have resulted in releases of chloroform to the land. It is impossible to quantify the magnitude of such releases, and the chloroform could be expected to be transported to either the atmosphere through volatilization or, if dissolved in water, carried into surface waters or groundwater.

5.3 ENVIRONMENTAL FATE

5.3.1 Transport and Partitioning

Based upon a vapor pressure of 159 mm Hg at 20 °C, chloroform is expected to exist almost entirely in the vapor phase in the atmosphere (Boublik et al. 1984; Eisenreich et al. 1981). Large amounts of chloroform in the atmosphere may be removed by wet deposition since chloroform has significant solubility in water. This is confirmed by its detection in rainwater (Kawamura and Kaplan 1983).

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Most of the chloroform removed in precipitation, however, is likely to reenter the atmosphere by volatilization. Trace amounts of chloroform have been documented in air samples from remote, often relatively pristine, areas of the world (Class and Ballschmidter 1986). Since chloroform is relatively nonreactive in the atmosphere, long-range transport within the atmosphere is possible. The detections in remote areas may also mean that the chloroform is produced as the result of more localized transformation processes, possibly including the reaction of naturally generated chlorinated oxidants with organic matter.

The dominant fate process for chloroform in surface waters is volatilization. Chloroform present in surface water is expected to volatilize rapidly to the atmosphere. An experimental half-disappearance range of 18-25 minutes has been measured for volatilization of chloroform from a 1 ppm solution with a depth of 6.5 cm that was stirred with a shallow pitch propeller at 200 rpm at 25 ºC under still air (≈0.2 mph air currents) (Dilling 1977; Dilling et al. 1975). Using the Henry's law constant, a halflife of 3.5 hours was calculated for volatilization from a model river 1 meter deep flowing at 1 meter/second, with a wind velocity of 3 m/second, and neglecting adsorption to sediment (Lyman et al. 1982). A half-life of 44 hours was estimated for volatilization from a model pond using EXAMS (1988).

Based on a measured soil organic carbon sorption coefficient (K_{oc}) of 45 (or a log [K_{oc}]of 1.65), chloroform is not expected to adsorb significantly to sediment or suspended organic matter in surface water (Sabljic 1984). This prediction is supported by sediment monitoring data that indicate that this compound has not been detected (or was detected at very low concentrations) in sediment samples (Bean et al. 1985; Ferrario et al. 1985; Helz and Hsu 1978). Little or no chloroform concentration was observed on peat moss, clay, dolomite limestone, or sand added to water (Dilling et al. 1975). Chloroform slightly adsorbed to aquifer solids in laboratory studies utilizing different amounts of two different aquifer materials with K_{oc} values ranging from 63.4 to 398. The authors reported higher adsorption with increasing organic content of the solids (Uchrin and Mangels 1986). K_{oc} values ranging from 45 to 80 in soil have been experimentally determined for chloroform (Sabljic 1984; Wilson et al. 1981).

Chloroform does not appear to bioconcentrate in higher aquatic organisms, based upon measured bioconcentration factors (BCF) of 6 and 8 for bluegill sunfish *(Lepomis macrochirus)* (Barrows et al. 1980; Veith et al. 1980). Information from EPA's ASTER (1996) database document a calculated

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BCF for the fathead minnow of 14, a low value suggesting little potential for bioconcentration in fish.
A BCF of 690 experimentally determined for the bioconcentration of chloroform in the green algae
Selenastrum capricornutum suggests that the compound has a moderate tendency to concentrate in
nonvascular aquatic plants (Mailhot 1987). No data regarding the biomagnification potential of
chloroform were found. Based upon the observed BCF, however, significant biomagnification of
chloroform is apparently unlikely.

In soil, the dominant transport mechanism for chloroform near the surface will probably be
volatilization because of its high volatility and low soil adsorption. Volatilization rates seem relatively
constant over a wide variety of soil types (Park et al. 1988). In other laboratory studies, 75% of the
chloroform initially present in water volatilized when applied to a fine sandy soil, and 54% of the
chloroform volatilized from a soil column during a percolation study utilizing a sandy soil (Piwoni et
al. 1986; Wilson et al. 1981). All or nearly all of the remaining chloroform traveled through the soil
because of its low adsorption onto soil. Another laboratory study of 15 common volatile or semivolatile
organic chemicals reported a disappearance half-life for chloroform of 4.1 days, which
assumed first-order kinetic decay (Anderson et al. 1991). The leaching potential of chloroform is
further confirmed by the detection of chloroform in groundwater, especially at hazardous waste sites
(Clark et al. 1982; Dewalle and Chian 1981; Harris et al. 1984; Sawhney 1989).

5.3.2 Transformation and Degradation

For air, the major degradation process involves reactions with free radicals such as hydroxyl groups
(Atkinson 1985). For other media, it is clear that chloroform can be mineralized through both abiotic
and biotic processes. Information in the available literature (Bouer and McCarty 1983; Rhee and
Speece 1992) documents the disappearance of chloroform in water and soil media under both aerobic
and anaerobic conditions as well as identification of the end products.

5.3.2.1 Air

The vapor-phase reaction of chloroform with photochemically generated hydroxyl radicals is the
dominant degradation process in the atmosphere. The rate constant for this process at 25 °C has been
experimentally determined as 1.0×10^{-13} cm^3/molecule-second, which corresponds to a half-life of
≈80 days based upon a 12-hour sunlit day in a typical atmosphere containing 1×10^6 hydroxyl

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radicals/cm^3 (Hampson 1980; Singh et al. 1981). Breakdown products from reaction with hydroxyl radicals probably include phosgene and hydrogen chloride (Atkinson 1985). Chloroform is more reactive in photochemical smog conditions where the approximate half-life is 11 days (Dimitriades and Joshi 1977). Direct photolysis of chloroform will not be a significant degradation process in the atmosphere. Chloroform solutions sealed in quartz tubes and exposed to sunlight for one year degraded at almost the same rate as solutions in sealed tubes stored in the dark, which indicated that very little or no photodegradation of the compound had occurred (Dilling et al. 1975). This is expected because chloroform does not show significant light absorbance at wavelengths >290 nm (Hubrich and Stuhl 1980).

5.3.2.2 Water

Hydrolysis will not be a significant degradation process in water based upon rate constants experimentally determined at 25 °C that correspond to half-lives ranging from 1,850 to 3,650 years at pH 7, and from 25 to 37 years at pH 9 (Jeffers et al. 1989; Mabey and Mill 1978). Direct photolysis of chloroform will not be a significant degradation process in surface waters because, as noted above, the compound does not absorb light at wavelengths >290 nm (Hubrich and Stuhl 1980). The reaction rate of chloroform with hydrated electrons photochemically produced from dissolved organic matter has been predicted to correspond to a near-surface half-life of ≈44 days based upon an experimentally determined rate constant and a hydrated electron concentration of 1.2×10^{-17} mol of hydrolyzed electrons/L (Zepp et al. 1987). This latter process is probably too slow to effectively compete with volatilization as a removal process from surface waters.

Biological degradation of chloroform has been studied primarily with an eye to batch process operation at waste water treatment plants or remediation possibilities at hazardous waste disposal sites. Above certain dosage levels, chloroform becomes toxic to anaerobic and aerobic microorganisms. This is especially noticeable for biological treatment facilities that use anaerobic digestion systems, where sustained inputs with chloroform concentrations approaching 100 mg/L can all but eliminate methanogenic (methane-fermenting) bacteria (Rhee and Speece 1992). Other studies have shown appreciable inhibition of methanogenesis with levels of chloroform in the range of 1 mg/L (Hickey et al. 1987). Other chlorinated hydrocarbons, and particularly such common 2-carbon chlorinated aliphatics as TCE, can similarly inhibit bacteria found in sewage sludges (Long et al. 1993; Rhee and Speece 1992). Similar inhibition effects can be the result of heavy metal toxics, zinc being

particularly stressful to methanogenic bacteria (van Beelen et al. 1994; van Vlaardingen and van Beelen 1992). Studies of actual natural waters or waste waters, where it is difficult to control the levels of specific chemicals or preclude inputs of other toxicants, yield a wide variety of results on the efficiencies of chloroform biodegradation. For instance, little or no degradation was observed during 25 weeks in aqueous aerobic screening tests utilizing primary sewage effluent inocula (Bouwer et al. 1981a). No chloroform degradation was observed in aerobic biofilm column studies (Bouwer et al. 1981b). Significant degradation of chloroform (46-49% loss in 7 days, at least some of which was apparently due to volatilization) in aerobic screening tests utilizing settled domestic waste water as inoculum was reported (Tabak et al. 1981). Under the proper conditions, chloroform appears to be much more susceptible to anaerobic biodegradation. Degradation of chloroform under anaerobic conditions was more rapid at lower chloroform concentrations (81 and 99% degradation after 2 and 16 weeks, respectively, at 16 ppb); a more gradual degradation was observed at higher concentrations and 78% degradation after 2 and 16 weeks, respectively, at 157 ppb) (Bouwer et al. 1981a). No degradation was observed, however, when chloroform was incubated with aquifer material under anaerobic conditions for 27 weeks (Wilson et al. 1981).

In the absence of toxicity from other solvents, chlorinated hydrocarbons, or heavy metals, and where chloroform concentrations can be held below approximately 100 ppb, both aerobic and anaerobic bacteria can biodegrade chloroform, with removal rates well over 80% in a period of 10 days (Long et al. 1993). Deviations from these ideal conditions can lead to lower removal efficiencies. These biodegradation reactions generally lead to the mineralization of the chloroform to chlorides and carbon dioxide (Bouwer and McCarty 1983; Rhee and Speece 1992). One study, however, documents the production of the toxicant methylene chloride (dichloromethane) from the breakdown of chloroformcontaining wastes in a mixed culture of bacteria from sewage sludge (Rhee and Speece 1992 citing results from work at Tyndall AFB, Florida). Caution should be exercised in making generalizations without site-specific evidence, however, since commercial grades of chloroform will often contain methylene chloride as an impurity (HSDB 1996). In waters containing mixtures of different chlorinated aliphatics, biodegradation may produce new chloroform, at least as a temporary by-product, the breakdown of carbon tetrachloride into chloroform having been confirmed in laboratory studies (ATSDR 1994; Long et al. 1993; Picardal et al. 1993).

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5.3.2.3 Sediment and Soil

Little information was located regarding the degradation of chloroform in soil. Based upon data for degradation in water, chemical degradation in soil is not expected to be significant. The available soil data suggest that chloroform biodegradation rates in soil may vary, depending upon conditions. In soil column studies, the chloroform present in the influent secondary waste water appeared to pass through the column nearly unchanged even though some of the other organic compounds present were apparently biodegraded, which indicated that the waste water was not too toxic to the microorganisms in the soil (Bouwer et al. 1981b). In contrast to these studies, significant degradation of chloroform (33% removed in 6 days) was observed in fine sandy soil in sealed bottles; however, the chloroform may have been co-metabolized by methylotropic bacteria already present in the soil. The aerobic degradation was even faster in methane-enriched soil (Henson et al. 1988). Such bio-oxidation of chloroform was also observed under methanogenic conditions in batch experiments using an inoculum derived from activated sludge and in a continuous-flow laboratory scale column, using a methanogenic fixed film derived from primary sewage effluent (Bouwer and McCarty 1983). Overall, biodegradation in soil is not expected to compete with the predicted rapid rate of volatilization from soil (Park et al. 1988). As with biodegradation in water, concentrations of chloroform above certain threshold levels may inhibit many bacteria, especially methane-fermenting bacteria under anaerobic or near-anaerobic conditions (Hickey et al. 1987).

5.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

5.4.1 Air

Data from the most recent study located (1982-85 air samples) reported that the background level of chloroform concentrations over the northern Atlantic ocean ranges from 2×10^{-5} to 5×10^{-5} ppm (Class and Ballschmidter 1986). This range does not differ significantly from the range reported for 1976-79 ($1.4\text{-}4 \times 10^{-5}$ ppm) and the range reported from the 1987 update of the National Ambient Volatile Organic Compounds Database (NAVOCDB), which was 2×10^{-5} ppm (Brodzinsky and Singh 1982; EPA 1988b; Singh 1977; Singh et al. 1979). The maximum and background levels found in 7 U.S. cities between 1980 and 1981 were 5.1×10^{-3} and 2×10^{-5} ppm, respectively (Singh et al. 1982). Average atmospheric levels in U.S. cities ranged from 2×10^{-5} to 2×10^{-3} ppm between 1980 and 1981, The median concentration reported between 1977 and 1980 was 7.2×10^{-5} ppm, and the median

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reported in the 1987 update of the NAVOCDB was 6×10^{-5} ppm (Brodzinsky and Singh 1982; EPA 1988b; Singh et al. 1981, 1982; Wallace et al. 1986a, 1986b, 1988). A recent update to EPA's volatile organic compounds (VOC) databases on background ambient air concentrations for hazardous air pollutants (Kelly et al. 1994) estimated a chloroform background ambient level for chloroform in the United States as 4×10^{-5} ppm.

The median concentration for source-dominated areas in the United States is 8.2×10^{-4} ppm for data reported between 1977 and 1980, and this figure does not differ significantly from the 5.1×10^{-4} ppm values reported in the 1987 update of the NAVOCDB (Brodzinsky and Singh 1982; EPA 1988b). Certain source-dominated areas contained much higher chloroform levels. The ambient air concentrations outside homes in Love Canal, New York, in 1978, ranged from 2×10^{-4} to 2.2×10^{-2} ppm, and the maximum concentration found in ambient air at 20 California municipal landfills was 0.61 ppm (Barkley et al. 1980; Wood and Porter 1987). Concentrations ranging from 2.9×10^{-4} to 6×10^{-3} ppm were found in air samples taken from 5 hazardous waste sites in New Jersey (LaRegina et al. 1986). Ambient air samples measured near a hazardous waste landfill contained $\leq 1 \times 10^{-3}$ ppm chloroform. All these data indicate that chloroform levels in air can be much higher in areas near hazardous waste sites (Stephens et al. 1986). Other source-dominated areas that may have ambient air chloroform concentrations significantly higher than background levels include areas near facilities that treat hazardous and municipal waste, as well as areas near contaminated groundwater, and municipal- and hazardous-waste incinerators (Corsi et al. 1987; EPA 1990a; LaRegina et al. 1986; Namkung and Rittmann 1987; Travis et al. 1986).

Typical median indoor air concentrations of chloroform range from $\approx 2 \times 10^{-4}$ to 4×10^{-3} ppm (Barkley et al. 1980; Pellizzari et al. 1986; Wallace et al. 1987c, 1989). Chloroform concentration ratios of indoor air to outdoor air range from <l to 25 (Pellizzari et al. 1986). One of the most significant indoor sources of chloroform is chlorinated tap water, and taking showers (and breathing air where chloroform has been released from the shower water) is expected to contribute a substantial amount to the indoor chloroform levels (Andelman 1985a, 1985b; Wallace 1987). A recent study investigating typical levels of various VOCs in the air in Canadian homes noted median chloroform concentrations of 4×10^{-4} ppm (Otson et al. 1994).

The air around swimming pools may also contain chloroform. This is especially likely in heated, indoor pools, which can approximate the conditions found in shower stalls. Concentrations ranging

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from 3.5×10^{-2} to 19.9×10^{-2} ppm (440-2,335 nmol/m^3) have been reported for environmental air at an indoor swimming pool (Aggazzotti et al. 1990, 1993).

Chloroform has been identified in at least 717 of 1,428 current or past NPL sites (HazDat 1996). Chloroform has been detected in air samples taken at an estimated 57 of the 717 NPL hazardous waste sites where chloroform was detected.

5.4.2 Water

Recent monitoring data regarding the presence of chloroform in surface water, sediments, and groundwater were not located. The most recent monitoring data that were obtained involved chloroform levels in drinking water. Finished drinking water collected in 1988 from 3.5 sites across the United States contained median concentrations of chloroform ranging from 9.6 to 15 µg/L (Krasner et al. 1989). In an analysis of available monitoring data from raw water supplies, the maximum detected value was 136 ppb (EPA/AMWA 1989). Data from earlier studies indicate a wide range of concentrations have been found in drinking-water supplies. The reported chloroform concentrations that were detected ranged from trace levels to 311 ppb, with one study reporting a median concentration of 16.7 ppb and another study reporting a geometric mean concentration of 1.81 ppb. Most of the concentrations ranged between 22 and 68 ppb (Brass et al. 1977; EPA/AMWA 1989; Furlong and D'Itri 1986; Kasso and Wells 1981; Krasner et al. 1989; Rogers et al. 1987; Symons et al. 1975). Chloroform can be expected to exist in virtually all chlorinated drinking-water supplies. The main source of chloroform found in municipal drinking water is the chlorination of naturally occurring humic materials found in raw-water supplies (Bellar et al. 1974; Cech et al. 1982). Factors that can increase the amount of chloroform in drinking water include seasonal effects (high summer values) and increased contact time between chlorine and humic material. Sources of water with high humic material content will contain higher levels of chloroform, The chloroform concentration increases with time, indicating that concentrations of the compound increase as the water moves through the distribution system (Kasso and Wells 1981). Drinking water derived from groundwater, especially groundwater at or near some hazardous waste sites and landfills, may contain higher levels of chloroform than normally encountered in drinking water derived from surface water. Chloroform levels ranging from 2.1 to 1,890 ppb have been observed in drinking water derived from wells near a hazardous waste dump (Clark et al. 1982). The leachate from one solid waste landfill contained 21,800 ppb chloroform; drinking water obtained from wells in the vicinity of the landfill had

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chloroform levels of 0.3-1.6 ppb (Dewalle and Chian 1981). Data from the most recent study of Kansas groundwater sampled in 1986 indicate concentrations ranging from ≈0.3 to 91 ppb in both raw and treated groundwater; the average and median concentrations in the treated water were 7.6 and 0.5 ppb, respectively (Miller et al. 1990). Of the sample sites in a national groundwater supply survey, 45% had detectable levels of chloroform, and median and maximum concentrations were 1.5 and 300 ppb, respectively (Westrick et al. 1989).

In addition to drinking water, chlorinated oxidants reacting with organic materials will lead to the formation of chloroform in swimming pools. Since swimming pools are not routinely analyzed for their chloroform content, data are limited and derived from special studies. Such studies will often cover the broader family of THMs, and while chloroform levels in blood or alveolar air samples will be reported, the environmental agents will often simply be recorded in terms of TI-IMs. A rule of thumb (Copaken 1990) is that up to 90% of the THMs in chlorinated water samples will be chloroform. Recorded concentrations in samples from public pools fall in a range of 25-137 ppb (Aiking et al. 1994; Barnes et al. 1989; Copaken 1990). In poorly tended or very crowded public or private pools, where there are large inputs of organic materials or heavy use of chlorinating agents, chloroform levels as high as 163 ppb have been documented (Barnes et al. 1989). Chloroform production in swimming pools can be increased where the pools are treated with copper-containing algicides. In tests on chlorinated water using various doses of chlorine, copper (cupric valence form) from different copper-containing salts, and varying levels of humic acid (Barnes et al. 1989), chloroform concentrations after a given reaction time were generally 50% or more higher in samples treated with copper, which acts as a catalyst in the reactions with the humic acids.

Current reviews of surface water monitoring data in the peer reviewed literature are lacking. The highest concentrations observed in surface waters of the United States sampled before 1984 were 394 and 120 ppb. These concentrations were observed in rivers in highly industrialized cities (Ewing et al. 1977; Pellizzari et al. 1979). Typical concentrations for most sites that are not heavily industrialized appear to range from trace levels to ≈22 ppb (Ohio River Valley Sanitation Commission 1980, 1982). Data from EPA's STORET database indicate that chloroform was detected in 64% of 11,928 surface water sample data points at a median concentration of 0.30 ppb (Staples et al. 1985).

Chloroform at 0.25 ppb has been found in rainwater collected in Los Angeles, California, during 1982 (Kawamura and Kaplan 1983).

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Chloroform has been detected in surface-water samples taken at an estimated 139 of 717 current or past NPL sites where chloroform was detected (HazDat 1996). Detections in groundwater are documented at 552 sites. From the available information found in HazDat (1996), groundwater appears to be the most common environmental medium at NPL sites where chloroform might be encountered.

5.4.3 Sediment and Soil

Chloroform has been found in sediment samples. Chloroform was found in sediment samples taken in 1980 from the 3 passes of Lake Pontchartrain, Louisiana, at concentrations ranging from 1.7 to 18 ng/kg (w/w [weight per weight] basis) (Ferrario et al. 1985). Chloroform was found at concentrations ranging from 30 to 80 ng/kg (dry weight basis) in sediment samples exposed to chlorinated electrical power plant cooling water; the control samples that were not exposed to cooling water contained nearly the same amounts of chloroform (Bean et al. 1985). Data from EPA's STORET database indicate that chloroform was detected in 8% of 425 sediment sample data points at a median concentration of <5.0 μg/kg (Staples et al. 1985).

Routine sediment sampling for chloroform does not seem common in the United States, and sampling at relatively unpolluted ambient monitoring sites may overlook the levels possible in more restricted hotspots. Anaerobic biodegradation of chlorinated hydrocarbons may generate chloroform, especially in harbors, shipping canals, or areas receiving large amounts of industrial discharges. Where the sediments contain large concentrations of zinc, which is widely used to galvanize metal or as an ingredient in common industrial rust inhibitors, methanogenic bacteria populations may be adversely affected, thus preventing the mineralization of chloroform. Chloroform in interstitial water may then build up to levels as high as 50 ppb (van Beelen et al. 1994; van Vlaardingen and van Beelen 1992).

Soil monitoring data in the peer reviewed literature could not be located. It can be predicted that chloroform contamination occurs at hazardous waste sites where chloroform-containing leachate moves through the soil to groundwater. An explanation of the lack of data results from the fact that any chloroform in the soil is expected to either rapidly volatilize or leach. Laboratory studies using a variety of different soil types document the effectiveness of volatilization in removing chloroform from soils (Park et al. 1988).

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Chloroform has been found in soil samples taken at an estimated 213 of the 717 current or past NPL where chloroform was detected (HazDat 1996). Detections in sediment samples are documented for 79 current or past NPL sites. Detections on soil gas obtained from soil samples were documented at 18 sites.

5.4.4 Other Environmental Media

Chloroform has been detected in various foods at the following concentrations: soft drinks and beverages (2.7-178 µg/kg), dairy products (7-1,110 µg/kg), oils and fats (traces <12 µg/kg), dried legumes (6.1-57.2 µg/kg), and grains and milled grain products (1.4-3,000 µg/kg) (Abdel-Rahman 1982; Entz et al. 1982; Graham and Robertson 1988; Heikes 1987; Heikes and Hopper 1986; Lovegren et al. 1979). In a study of various foods, 41% of 231 samples contained chloroform at levels ranging from 4 to 312 µg/kg; the average level was 52 µg/kg (Daft 1988a). In another broad study, 55% of 549 samples contained between 2 and 830 µg/kg. The average level in this study was 71 µg/kg (Daft 1989). The chloroform concentration observed in other foods ranged from 6.1 to 1,110 µg/kg. The highest amounts were found in butter (1,110 µg/kg), mixed cereal (220µg/kg), infant/junior food (230 µg/kg), and cheddar cheese (83 µg/kg) (Heikes 1987).

Chloroform has been detected in the air above outdoor and indoor pools and in spas at maximum concentrations of 2.8×10^{-2}, 5.0×10^{-2}, and 5.2×10^{-2} ppm, respectively; water concentrations ranged between 4 and 402, 3 and 580, and <0.1 and 530 ppb, respectively (Armstrong and Golden 1986). In another study, air samples above whirlpool spas treated with chlorine disinfectant contained chlorofom at concentrations ranging from 8×10^{-4} to 1.5×10^{-1} ppm; the concentration in the water ranged from 15 to 674 ppb (Benoit and Jackson 1987). Chloroform has been detected at ≤37 ppb in the cooling water of a nuclear reactor; a concentration of 50 ppb was detected 0.75 miles downstream from the reactor cooling tower in one study (Hollod and Wilde 1982).

Since chloroform is highly volatile and shows little tendency to bioconcentrate or bioaccumulate in higher life forms such as fishes, it is not ordinarily included in the types of persistent pollutants that are the focus of state fish consumption advisory programs. Information from HazDat (1996) does document detections of chloroform in tissues from fishes from at least 3 current or past NPL sites.

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5.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

The general population is probably exposed to chloroform through drinking water and beverages, eating food, inhaling contaminated air, and through dermal contact with water (e.g., while showering, bathing, cleaning, washing, swimming). All humans are expected to be exposed to at least low levels of chloroform. Accurate, current estimates of the daily intake of chloroform by various exposure routes are not available, or possible, due to the lack of appropriate current monitoring data. Typical levels of atmospheric exposure in remote, urban, and source-dominated areas range from $2x10^{-5}$ to $5x10^{-5}$, $6x10^{-5}$ to $2x10^{-3}$, and $8.2x10^{-4}$ to $2.2x10^{-2}$ ppm, respectively (Barkley et al. 1980; Brodzinsky and Singh 1982; Class and Ballschmidter 1986; EPA 1988a; Singh et al. 1981, 1982; Wallace et al. 1986a, 1986b, 1988; Wood and Porter 1987). Exposure via ingestion of contaminated drinking water is expected to be extensive since most U.S. community drinking-water supplies are chlorinated (see Singer 1994 for an overview of the entire topic of disinfection by-products in drinking water). Typical levels in drinking water range from 2 to 68 ppb (Brass et al. 1977; EPA/AMWA 1989; Furlong and D'Itri 1986; Kasso and Wells 1981; Krasner et al. 1989; Rogers et al. 1987; Symons et al. 1975). Although data regarding levels in food are rather scant, typical average chloroform levels in certain foods are estimated to range from 52 to 71 μg/kg (Daft 1988a, 1988b, 1989).

Although data are available from various studies regarding concentrations of chloroform found in human tissues, blood, and expired air, only limited data are available that compare these concentrations to measured or estimated environmental exposure levels. Furthermore, no correlation has been made between these measured human tissue concentrations and the corresponding environmental exposure levels (see Section 2.5.1 for a discussion of the relationship between chloroform exposure levels and concentrations found in humans). Much of the data available is from the Total Exposure Assessment Methodology (TEAM) studies (see Andelman 1990 or Wallace 1995 for succinct overviews) in which the concentration of chloroform was measured in personal air samples and exhaled human breath (Wallace 1987; Wallace et al. 1984, 1986a, 1986b, 1988). For example, in one TEAM study, the ratios of the concentrations of chloroform detected in personal air samples to those found in human exhaled breath air varied from 0.66:1 to 13.3:1 (Wallace 1987) (see Section 2.5.1 for more data regarding concentrations found in humans, including data obtained during autopsies). Recent studies supported through IARC on swimmers using indoor pools in Europe showed good correlations between the chloroform concentrations in alveolar air and blood plasma concentrations (Aggazzotti et al. 1990, 1993). Plasma levels ranging from 0.8 to 25.1 nmol/L were observed. Statistical analysis

showed the plasma levels to be significantly correlated with concentrations of chloroform in the pool water, time spent swimming, the number of swimmers in the pool, and the chloroform concentration in the environmental air.

Limited current data were located regarding occupational exposure to chloroform. Although some of the exposure levels encountered in workplaces may be comparable to exposure the worker receives in his own home, there are probably many specific jobs that expose the workers to significantly higher levels of chloroform. These occupations include work at or near source-dominated areas such as chemical plants and other facilities that manufacture or use chloroform, operation of chlorination processes in drinking-water plants, work at or near waste water-treatment plants and paper and pulp plants, and other facilities where large amounts of chloroform are released (e.g., hazardous and municipal-waste incinerators). Persons working at waste water and other treatment plants can be exposed to significant levels of chloroform. A maximum level of 3.8×10^{-3} ppm was found in the air at an activated sludge waste-water treatment plant (Lurker et al. 1983). Maintenance workers, attendants, and life guards at indoor pools and spas may encounter maximum concentrations of 5.0×10^{-2} and 1.5×10^{-1} ppm, respectively (Armstrong and Golden 1986; Benoit and Jackson 1987). Persons who use tap water often, especially if it is heated and/or sprayed (e.g., water used for cleaning, washing clothes and dishes, showering, and cooking), may be exposed to higher than background levels. For example, levels in personal air samples as high as 2.2×10^{-2} and 1.1×10^{-2} ppm have been measured during household cleaning activities and showering (Wallace et al. 1987d). While the use of activated carbon filters may provide some reduction in the tap water levels for cold water feeds, such filters are not effective with hot water where the elevated temperatures will induce volatilization from the filter media. Persons using certain cleaning agents and pesticides in enclosed spaces with poor ventilation or persons working where these materials are used may be exposed to relatively high levels of chloroform.

A National Occupational Exposure Survey (NOES) conducted by NIOSH from 1981 to 1983 estimated that 95,778 workers in the United States are potentially exposed to chloroform (NOES 1991). In the absence of more recent national occupational exposure data, the NOES information still has some value. The NOES database does not contain information on the frequency, concentration, or exposure duration of workers; it only provides estimates of the number of workers potentially exposed to chemicals in the workplace. Of the 151 different occupational groups with potential chloroform exposure risks listed in the NOES database, approximately 50% of the potentially exposed workers fall

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into the following Standard Industrial Classification (SIC) categories: Funeral Service and Crematory Technicians from SIC category 7261; Biological, Engineering or Chemical Technicians and Chemists from the Research and Development Laboratory SIC category 7391; Adhesive and Sealant Chemical Technicians from SIC category 2891; Assemblers from SIC category 3679 (Electronic Components) and SIC Category 3622 (Industrial Controls); Petroleum Refining Machine Operators and other Workers from SIC category 2911; and General Medical and Surgical Hospital Clinical Laboratory Technologists and Technicians from SIC category 8062.

5.6 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

All humans are exposed to low concentrations of chloroform. Those with potentially high exposures are workers employed in chloroform manufacturing and use industries. Persons living in certain source-dominated areas may be at risk for higher than background exposures to chloroform. These may include persons living near industries and facilities that manufacture and use chloroform, municipal and industrial waste water-treatment plants and incinerators, paper and pulp plants, and persons who derive their drinking water from groundwater sources contaminated with chloroformcontaining leachate from hazardous waste sites.

Previously reported air monitoring data from landfills and other waste sites (see Section 5.4.1) suggest that potentially high exposure may occur via inhalation of contaminated air near hazardous waste sites. Other possibilities include drinking water from wells contaminated with chloroform that leached from the sites and, perhaps, living in homes built directly on top of former waste sites. Although some of the drinking-water supplies contaminated solely by leached chloroform (e.g., levels ≤1.6 ppb from a water well near a site with documented chloroform contamination) (Dewalle and Chian 1981) have shown levels lower or comparable to that in normal chlorinated drinking water, where chloroform levels in the range of 2-44 ppb are common (EPA/AMWA 1989), much higher levels (1,890 ppb) have been found in water from wells near a waste dump (Clark et al. 1982). HazDat (1996) documents 239 current or past NPL sites where specific concentration levels for groundwater are contained in the database. At least 97 current or past NPL sites show groundwater samples where values are 100 ppb or higher. A drinking-water standard of 100 ppb for total trihalomethanes (TTHMs) (where chloroform is usually the predominant constituent) is EPA's current requirement for treated water systems serving 10,000 or more consumers (see Chapter 7 below).

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5.7 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 chloroform is available. Where adequate information is not available, ATSDR, in conjunction with the 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 chloroform.

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.

5.7.1 Identification of Data Needs

Physical and Chemical Properties. As reported in Table 3-2, the physical and chemical properties of chloroform have been characterized sufficiently to permit estimation of its environmental fate.

Production, Import/Export, Use, Release, and Disposal. Data regarding the production methods and current, past, projected future production capacity volumes, and current import and export volumes are available (Ahlstrom and Steele 1979; CEN 1995; Deshon 1979; NTDB 1994; SRI 1993, 1994; TR192 1994). However, these statistics will generally not include all instances where chloroform is generated as a chemical intermediate or waste product. With the exception of the partial coverage provided in the Toxics Release Inventory (TR191 1993; TR193 1995) comprehensive information regarding current release and disposal patterns, are lacking. General disposal information is adequately detailed in the literature, and information regarding disposal regulations of chloroform is available (EPA 1988a, 1988b). Production, release, and disposal data are useful to determine where environmental exposure to chloroform may be high. A major data need is to achieve a better understanding of why a substantial number of NPL sites show chloroform levels in groundwater at or above concentrations of 100 ppb. There are 239 sites in HazDat (1996) where groundwater

concentration levels are documented. At least 36 sites show signs of groundwater contamination with levels of 1,000 ppb (or 1 ppm) or higher, levels high enough to suggest disposal or spills of wastes containing very high concentrations of chloroform. Sixty-one (61) sites show groundwater levels in a range from 100 to 1,000 ppb, and the remaining 142 sites show levels below 100 ppb. While some sites show very low levels comparable to the concentrations associated with many chlorinated public drinking-water supplies, the substantial number of sites with values at or slightly above the community drinking-water standard for TTHMs of 100 ppb deserves some scrutiny. Is the chloroform at such sites the result of past disposal of chloroform-laden wastes, or are some other sources (e.g., *in situ* generation of chloroform as the result of chemical or biochemical transformation of other on-site wastes) for the chloroform contamination involved? Present data sources are lacking to explain adequately the large number of NPL sites showing elevated levels of chloroform in such media as groundwater that pose major off-site exposure risks.

Environmental Fate. Experimental data are available regarding the transport and partitioning properties of chloroform in surface waters (Bean et al. 1985; Clark et al. 1982; Class and Ballschmidter 1986; Dilling 1977; Ferrario et al. 1985; Piwoni et al. 1986; Sawhney 1989). Chloroform partitions mainly into the atmosphere and into groundwater. Empirical measurements or model predictions on half-disappearance times in such media as soil could not be identified in the literature. Chloroform can be transported long distances in air. Data are available regarding the degradation of chloroform in the atmosphere, but less is known about degradation rates in water and soil (Anderson et al. 1991; Bouwer et al. 1981a, 1981b; Dilling et al. 1975; Hampson 1980; Henson et al. 1988; Jeffers et al. 1989; Park et al. 1988; Singh et al. 1981; Tabak et al. 1981; Wilson et al. 1981). Hydrolysis and direct photodegradation are not significant removal processes. Although data regarding biodegradation rates in natural media are lacking, volatilization is expected to dominate over biodegradation as a removal process from surface water and near-surface soil. Chloroform seems relatively persistent in the atmosphere and groundwater. The environmental fate of chloroform releases related to most common anthropogenic sources appears to be sufficiently determined by the available data. In light of the documented occurrence (Class and Ballschmidter 1986) of chloroform in remote, often pristine, areas, however, further study is warranted to help quantify the relative role of long-range transport processes as opposed to a variety of more localized potential chemical transformation processes. These more localized processes could include the reaction of naturally generated chlorinated oxidants with organic materials to yield chloroform. More data would be useful on the half-lives of chloroform in such media as soils.

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Bioavailability from Environmental Media. Chloroform is absorbed following inhalation, oral, and dermal contact. Toxicity studies of exposure to chloroform in air, water, and food demonstrated the bioavailability of chloroform by these routes. Data regarding its bioavailability from soil are lacking, but near-surface soil concentrations can be expected to be low due to volatilization (Piwoni et al. 1986; Wilson et al. 1981).

Food Chain Bioaccumulation. Data are available that indicate that chloroform does not bioconcentrate in aquatic organisms (Barrows et al. 1980; Veith et al. 1980); however, data are lacking for plants and other animals (e.g., vacuolar plants, shellfish, or macroinvertebrates) as well as for the biomagnification potential of chloroform in terrestrial and aquatic food chains. Additional information on bioconcentration and biomagnification could be useful in establishing the significance of food chain bioaccumulation as a route of human exposure.

Exposure Levels in Environmental Media. All humans are exposed to at least low levels of chloroform via inhalation of contaminated air, and most humans are exposed by drinking contaminated water. Estimates from intake via inhalation and ingestion of drinking water, based on limited data, are available (see Section 5.5). Exposure from foods cannot be estimated, due to the lack of data. Current information on exposure to chloroform from water, air, and foods, especially for workers or people who live near manufacturing and use facilities, water and waste water-treatment plants, municipal and industrial incinerators, hazardous waste sites, and other sources of significant release, in addition to data regarding exposure levels in indoor air would be useful.

Exposure Levels in Humans. Data regarding exposure levels in humans are incomplete and are usually the result of limited, special studies. Chloroform has been found in human blood and expired air of both occupationally and nonoccupationally exposed groups, and in breast milk of nonoccupationally exposed groups (Hajimiragha et al. 1986; Pellizzati et al. 1982; Wallace et al. 1987a). A detailed recent database of exposure would be helpful in determining the current exposure levels, thus allowing an estimation of the average daily dose associated with various scenarios, such as living near a point source of release, drinking contaminated water, or working in a contaminated place. This information is necessary for assessing the need to conduct health studies on these populations.

Exposure Registries. No exposure registries for chloroform were located. This substance is not currently one of the compounds for which a subregistry has been established in the National Exposure

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Registry. The substance will be considered in the future when chemical selection is made for subregistries 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.

5.7.2 Ongoing Studies

As part of the Third National Health and Nutrition Evaluation Survey (NHANES III), the Environment Health Laboratory Sciences Division of the National Center for Environment Health, Centers for Disease Control and Prevention, will be analyzing human urine samples for chloroform (Needham et al. 1990). These data will give an indication of the frequency of occurrence and background levels of these compounds in the general population.

Research at Colorado State University (FEDRIP 1994) will investigate toxicity and bioaccumulation potential of chloroform and other organic and inorganic chemicals. The laboratory studies will use a simple food chain containing larval invertebrates and fish. The results would be applicable to many natural systems with fish foraging on insect larvae in contaminated sediments.

Another research project at Colorado State University (FEDRIP 1994) will study the microbial degradation kinetics of pollutant mixtures, which will include chloroform. One aspect of this research will focus on both the degradation of chloroform as well as its inhibitory effects when present above certain threshold concentrations.

A project at the University of Arizona (FEDRIP 1996) will study microbial dehalogenation of several compounds, including chloroform. A major part of the study will focus on the facultative anaerobic bacteria *Shewanellu putrefaciens* sp., which is known to catalyze the transformation of carbon tetrachloride to chloroform and other as yet unidentified products. The organic substrates will also contain metals. It is hoped that the end-products from the biochemical treatment can be subjected to a photolytic finishing process that will completely mineralize any remaining halogenated compounds.

A project at the University of Idaho (FEDRIP 1996) will study the biodegradation of several halogenated hydrocarbons in soils amended with plant residues from various Brussicu cultivars (e.g., rape seed). These residues contain chemicals that may help catalyze the degradation of some

chlorinated hydrocarbons. Work to date has shown the potential for phrophyrins metallated with cobalt or nickel to cause the breakdown of carbon tetrachloride into such products as chloroform, dichlormethane, and carbon monoxide.

A project at the University of California at Berkeley (FEDRIP 1996) will study factors affecting the rates of degradation for several chlorinated hydrocarbons by soil microbial populations. Chloroform will be included as one of the organic chemicals studied.

A project conducted by the USDA (FEDRIP 1996) at its Western Regional Research Center in Albany, California, will study the use of ozone or peroxide treatment as an alternative to chlorinated bleaching in the preparation of walnuts intended for in-the-shell sales. There are concerns that the current use of hypochlorite bleaching agents can adversely impact taste as well as leaving behind THM residues containing chloroform.

A project at the University of California at Riverside (FEDRIP 1996) will study factors affecting the biodegradation in soils of several pesticides and halogenated organics by such microbes as *Methanobacterium thermoautotrophicum.* This anaerobic bacterium shows the potential for very rapid oxidation of several organics, including chloroform. In practice, however, one or more limiting factors dramatically reduces the expected degradation kinetics.

A project at the USDA's Western Research Center in Albany, California, will study ways to reduce exposures to chloroform for workers at poultry processing plants (FEDRIP 1994). At various points in the processing of poultry, the carcasses are rinsed in chiller-water baths that kill pathogens. Currently, chlorine is the only USDA-approved sanitizing agent. This study will investigate the potential of such alternative agents as chlorine dioxide to achieve comparable germicidal effects while reducing the levels of chloroform generated.

The United States Geological Survey (FEDRIP 1996) is conducting a study on the Mississippi River and its major tributaries focused on the transport and degradation of organic substances. Experiments will be conducted at 12 stations to determine the THM and organic halide formation potentials. The results will be summarized in a chapter on water purification by-products in a Report to Congress on the Mississippi River.

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Remedial investigations and feasibility studies at NPL sites that contain chloroform will also provide further information on environmental concentrations and human exposure levels near waste sites.