

## 5. POTENTIAL FOR HUMAN EXPOSURE

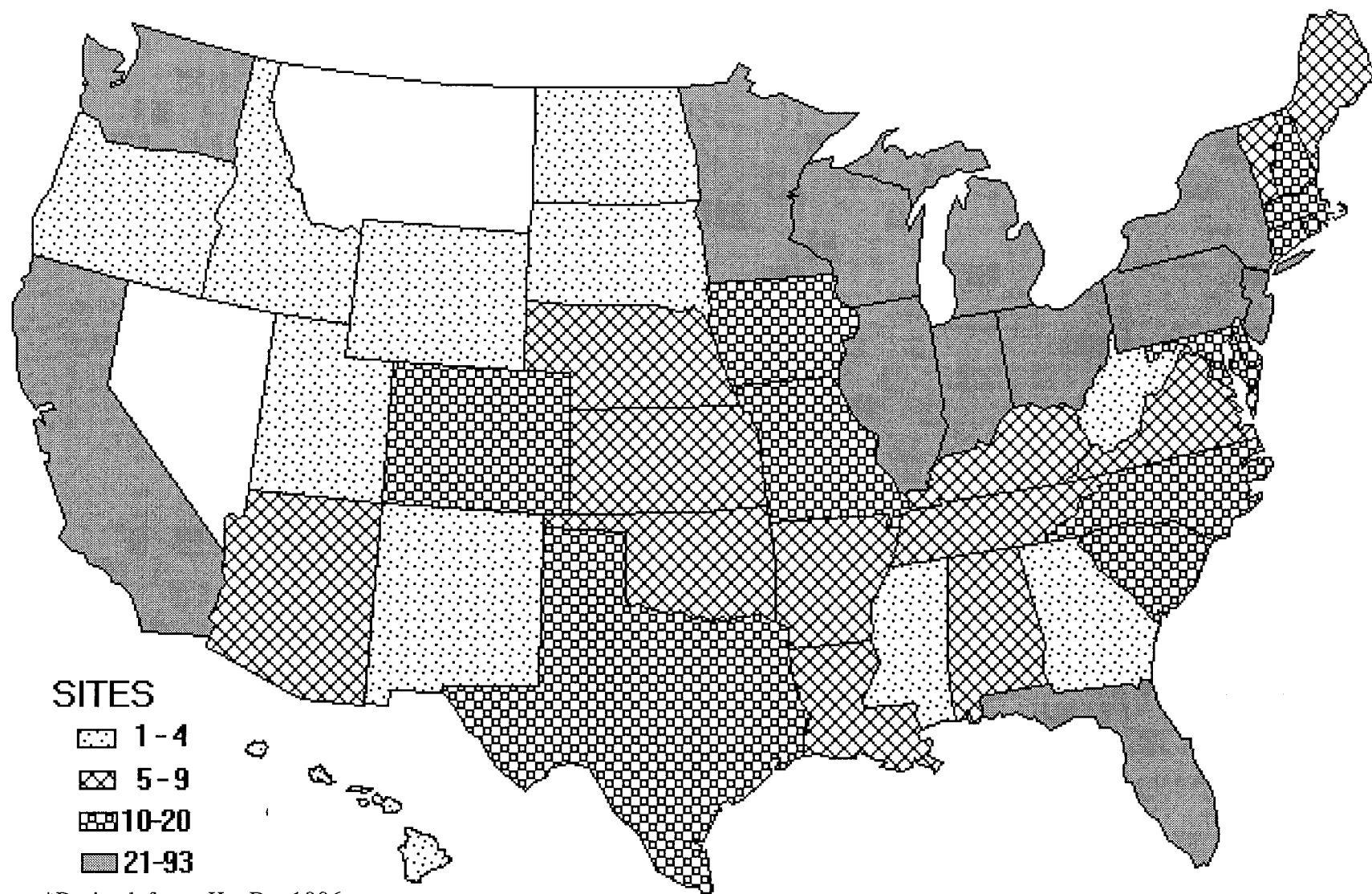
### 5.1 OVERVIEW

Trichloroethylene has been identified in at least 861 of the 1,428 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 1996). However, the number of sites evaluated for trichloroethylene is not known. The frequency of these sites can be seen in Figure 5-1. Of these sites, 857 are located in the United States and 3 are located in the Commonwealth of Puerto Rico and 1 in the Virgin Islands (not shown).

Most of the trichloroethylene used in the United States is released into the atmosphere by evaporation primarily from degreasing operations. Once in the atmosphere, the dominant trichloroethylene degradation process is reaction with hydroxyl radicals; the estimated half-life for this process is approximately 7 days. This relatively short half-life indicates that trichloroethylene is not a persistent atmospheric compound. Most trichloroethylene deposited in surface waters or on soil surfaces volatilizes into the atmosphere, although its high mobility in soil may result in substantial percolation to subsurface regions before volatilization can occur. In these subsurface environments, trichloroethylene is only slowly degraded and may be relatively persistent.

In general, atmospheric levels are highest in areas of concentrated industry and population and lower in rural and remote regions. Workers, particularly in the degreasing industry, are exposed by inhalation to the highest levels of trichloroethylene. Based upon monitoring surveys, these workers may be exposed to levels ranging from approximately 1 to 100 ppm. The general population can also be exposed to trichloroethylene by contact with and/or consumption of water from supplies contaminated with the chemical, by consumption of contaminated foods, and by contact with consumer products containing the compound. Based on available federal and state surveys, between 9% and 34% of the drinking water supply sources that have been tested in the United States may have some trichloroethylene contamination. It should be noted that the amount of trichloroethylene found by chemical analysis is not necessarily the amount that is bioavailable.

FIGURE 5-1. FREQUENCY OF NPL SITES WITH TRICHLOROETHYLENE CONTAMINATION\*



\*Derived from HazDat 1996

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**5.2 RELEASES TO THE ENVIRONMENT****5.2.1 Air**

According to the Toxic Chemical Release Inventory database (TRI), an estimated total of at least 49 million pounds of trichloroethylene was released to air from manufacturing and processing facilities in the United States in 1988 (TRISS 1990). The level reported in 1993 was 30.2 million pounds (TR1931995). The number of reporting facilities in each state and the ranges within which individual facilities reported their releases are shown in Table 5-1. The TRI data listed in this table should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list.

In a comprehensive study of trichloroethylene emission sources from industry conducted for EPA, the major source was degreasing operations, which eventually release most of the trichloroethylene used in this application to the atmosphere (EPA 1985e). Degreasing operations represented the largest source category of trichloroethylene emissions in 1983, accounting for about 91% of total trichloroethylene emissions. Other emission sources include relatively minor releases from trichloroethylene manufacture, manufacture of other chemicals (similar chlorinated hydrocarbons and polyvinyl-chloride), and solvent evaporation losses from adhesives, paints, coatings, and miscellaneous uses.

A recently discovered natural source of trichloroethylene is its production by several species of marine macroalgae and at least one species of marine microalgae (Abrahamsson et al. 1995). Rates of production ranged from 0.022 to 3400 ng/g fresh weight/hour, with the higher rates seen in subtropical *Rhodophyta* species. The importance of this source of trichloroethylene cannot be estimated at this time due to the lack of knowledge of its production in other species of algae. Also not fully understood is the physiology of how trichloroethylene is produced and how environmental factors may affect its production rate. There are too many unknown factors to determine whether this source could be a potential concern as a major source of atmospheric emissions of trichloroethylene in coastal areas.

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

State <sup>b</sup>	Number of Facilities	Reported amounts released in pounds per year <sup>a</sup>						
		Air	Water	Land	Underground injection	Total environment <sup>c</sup>	POTW	Off-site waste transfer
AL	12	1,500-424,000	0-250	0	0	1,500-424,005	0-209	0-186,104
AR	11	2,686-235,200	0	0	0	2,686-235,200	0-5	0-54,176
AZ	4	15,450-55,140	0	0	0	15,450-55,140	0	250-10,200
CA	5	5-13,936	0-1	0	0	5-13,936	0-9	0-3,261
CO	1	6,000	0	0	0	6,000	0	4,000
CT	15	3-94,000	0-148	0	0	3-94,000	0	0-90,999
DE	1	240,000	0	0	0	240,000	0	3,400
FL	14	0-126,000	0	0	0	0-126,000	0-5	0-16,968
GA	12	400-330,303	0	0-750	0	400-330,303	0-550	0-124,827
IA	5	1,363-15,446	0	0	0	1,363-15,446	0-5	0-19,138
IL	99	0-342,748	0-3	0-5	0	0-342,748	0-27,075	0-139,000
IN	50	0-1,126,000	0-250	0	0	0-1,126,000	0-300	0-105,100
KS	12	2,079-514,490	0-5	0	0-460	2,539-514,495	0-5	0-44,345
KY	17	0-216,600	0-5	0	0	0-216,600	0-5	0-39,462
LA	11	3-63,889	0-2,900	0-250	0	3-63,889	0	0-97,000
MA	36	10-176,200	0-5	0	0	10-176,200	0-5	0-248,300
MD	4	9,680-73,100	0	0	0	9,680-73,100	0-2,795	250-20,020
ME	2	8,167-75,375	0	0	0	8,167-75,375	0	0-60,997
MI	41	0-533,400	0-7	0	0	0-533,400	0-250	0-153,206
MN	28	2,640-154,220	0	0	0	2,640-154,220	0-567	0-50,160
MO	28	0-93,620	0-1	0	0	0-93,620	0-250	0-230,653
MS	6	8,900-1,219,813	0	0	0	8,900-1,219,813	0	1,233-12,600
NC	19	0-171,600	0	0	0	0-171,600	0-250	0-3,155,000

Table 5-1 (continued)

State <sup>b</sup>	Number of Facilities	Reported amounts released in pounds per year <sup>a</sup>						
		Air	Water	Land	Underground injection	Total environment <sup>c</sup>	POTW	Off-site waste transfer
NE	10	10,639-138,585	0	0	0	10,639-138,585	0-1	0-17,729
NH	4	2,810-88,000	0	0	0	2,810-88,000	0-1	9,021-56,000
NJ	11	0-241,310	0	0	0	0-241,310	0-650	0-120,900
NY	55	4-261,000	0-250	0-600	0	4-261,000	0-9,000	0-66,407
OH	55	0-332,000	0-250	0-6,600	0	0-332,250	0-250	0-209,500
OK	5	14,000-56,005	0	0	0	14,000-56,005	0	0-17,000
OR	7	14,605-186,000	0	0	0	14,605-186,000	0	45-9,548
PA	54	0-201,208	0-500	0-5	0	0-201,713	0-5	0-206,629
PR	1	12,000	0	0	0	12,000	0	6,022
RI	5	500-28,250	0	0	0	500-28,250	0-5	0-32,069
SC	12	500-130,750	0	0	0	500-130,750	0-43	0-150,000
SD	2	255-4,000	0	0	0	255-4,000	0	11,000-11,100
TN	14	6-122,720	0-5	0	0	6-122,720	0-250	0-28,330
TX	34	0-117,683	0-20	0-1	0	0-117,683	0-630	0-40,000
VA	12	2,111-320,255	0	0	0	2,111-320,255	0-250	110-92,585
VT	3	1-1,980	0	0	0	1-1,980	0	1-43,340
WA	9	0-36,278	0	0	0	0-36,278	0-5	0-37,150
WI	42	0-273,300	0-2	0	0	0-273,300	0-5	0-183,590

Source: TRI93 1995

<sup>a</sup>Data in TRI are maximum amounts released by each facility.<sup>b</sup>Post office state abbreviations used<sup>c</sup>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

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Release of trichloroethylene also occurs at treatment and disposal sites. Water treatment facilities may release trichloroethylene from contaminated water through volatilization and air-stripping procedures (EPA 1985e). Trichloroethylene is also released to the atmosphere through gaseous emissions from landfills. The compound may occur as either an original contaminant or as a result of the decomposition of tetrachloroethylene. Trichloroethylene has also been detected in stack emissions from the incineration of municipal and hazardous waste (James et al. 1985; Oppelt 1987).

### 5.2.2 Water

According to TRI, an estimated total of at least 13,800 pounds of trichloroethylene was released to water from manufacturing and processing facilities in the United States in 1988 (TRIM 1990). The level reported in 1993 was 5,468 pounds (TR1931995). The number of reporting facilities in each state and the ranges within which individual facilities reported their releases are shown in Table 5-1. The TRI data listed in this table should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list.

Trichloroethylene is released to aquatic systems from industrial discharges of waste water streams (EPA 1985c). Various monitoring studies nationwide have also found that trichloroethylene from landfill leachate can contaminate groundwater (DeWalle and Chian 1981; Kosson et al. 1985; Reinhard et al. 1984; Sabel and Clark 1984; Schultz and Kjeldsen 1986). In fact, trichloroethylene is the most frequently reported organic contaminant in groundwater (Bourg et al. 1992).

### 5.2.3 Soil

According to TRI, an estimated total of at least 21,190 pounds of trichloroethylene was released to land from manufacturing and processing facilities in the United States in 1988 (TRISS 1990). The level reported in 1993 was 8,213 pounds (TR1931995). The number of reporting facilities in each state and the ranges within which individual facilities reported their releases are shown in Table 5-1. The TRI data listed in this table should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list.

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Trichloroethylene can be released into the soil through industrial discharges into surface waters and through landfill leachate. EPA regulations now restrict the disposal of hazardous waste containing greater than or equal to 1,000 mg/kg halogenated organic compounds (such as trichloroethylene) in landfills (EPA 1987e).

### 5.3 ENVIRONMENTAL FATE

#### 5.3.1 Transport and Partitioning

The relatively short predicted half-life of trichloroethylene in the atmosphere indicates that long-range global transport is unlikely (Class and Ballschmiter 1986). However, its constant release, as well as its role as an intermediate in tetrachloroethylene degradation, may account for its persistence and the fact that trichloroethylene is often present in remote areas.

Trichloroethylene has been detected in a number of rainwater samples collected in the United States and elsewhere (see Section 5.4.2). It is moderately soluble in water, and experimental data have shown that scavenging by rainwater occurs rapidly (Jung et al. 1992). Trichloroethylene can, however, be expected to revolatilize back to the atmosphere after being deposited by wet deposition. Evaporation from dry surfaces can also be predicted from the high vapor pressure.

The Henry's law constant value of  $2.0 \times 10^{-2}$  atm- $m^3$ /mol at 20°C suggests that trichloroethylene partitions rapidly to the atmosphere from surface water. The major route of removal of trichloroethylene from water is volatilization (EPA 1987). Laboratory studies have demonstrated that trichloroethylene volatilizes rapidly from water (Chodola et al. 1989; Dilling 1977; Okouchi 1986; Roberts and Dandliker 1983). Dilling et al. (1975) reported the experimental half-life with respect to volatilization of 1 mg/L trichloroethylene from water to be an average of 21 minutes at approximately 25°C in an open container. Although volatilization is rapid, actual volatilization rates are dependent upon temperature, water movement and depth, associated air movement, and other factors. A mathematical model based on Fick's diffusion law has been developed to describe trichloroethylene volatilization from quiescent water, and the rate constant was found to be inversely proportional to the square of the water depth (Peng et al. 1994).

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Mathematical modeling of trichloroethylene volatilization from a rapidly moving, shallow river (1 meter deep, flowing 1 meter per second, with a wind velocity of 3 meters per second) has estimated its half-life at 3.4 hours (Thomas 1982). Measured volatilization half-lives in a mesocosm, which simulated the Narragansett Bay in Rhode Island during winter, spring, and summer, ranged from 13 days in summer conditions to 28 days in spring conditions (Wakeham et al. 1983).

Volatilization of trichloroethylene from soil is slower than it is from water but more rapid than that of many other volatile organic compounds (Park et al. 1988). This study found that an average of 37% of the applied trichloroethylene was volatilized 168 hours after treatment at 12°C and 45% was volatilized at 21°C. This study also concluded that soil type had no effect on rate of volatilization, although this may simply be a reflection of the fact that the differences between soils used in the study, particularly in organic carbon content, were not very great.

Sorption of organic compounds to soil has been found to be most reliably predicted when related to the organic carbon content of the soil (Kenaga 1980; Urano and Murata 1985). Experimentally measured soil organic carbon sorption coefficients ( $K_{oc}$  values) for trichloroethylene range from 106 to 460 (Garbarini and Lion 1986). The components of soil organic matter had widely varying affinities for trichloroethylene, with the fats-waxes-resins fraction ( $K_{oc} = 460$ ) being responsible for stronger adsorption of trichloroethylene. The calculated  $K_{oc}$  values are indicative of medium-to-high mobility in soil (Kenaga 1980; Swann et al. 1983). Others have also shown that trichloroethylene is highly mobile in sandy soil (Wilson et al. 1981). Another study comparing predicted and observed sorption on clay and organic soils suggested that sorption/desorption to inorganic mineral surfaces may also play a role, and the reactions generally follow reversible pseudo first-order kinetics (Doust and Huang 1992).

Several models for describing the transport of volatile chlorinated hydrocarbons in soils have been developed, often by fitting one or more parameters to experimental data. One model which determined all parameters a priori and included transfer between solid, liquid, and gas phases found that the Henry's law constant was the primary determinant of transport behavior in a wet nonsorbing aggregated medium, suggesting that volatilization and movement in the gas phase accounts for a large portion of trichloroethylene movement in soils (Gimmi et al. 1993). However, as the velocities of the gas and liquid phases increase, equilibrium partitioning is less likely, and prediction from Henry's law



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is less reliable. This was found to be the case in laboratory and field experiments on trichloroethylene volatilization from contaminated groundwater and diffusion through soil (Cho et al. 1993). In addition, sorption of trichloroethylene to the surfaces of soil particles, which may decrease its transport and bioavailability, is dependent on soil moisture content, since polar water molecules will compete aggressively with nonpolar vapor phase trichloroethylene for polar sorption sites. This has been experimentally confirmed with real soil samples, in which it was found that the solid/vapor partition coefficient decreased dramatically with increased moisture content (Peterson et al. 1994).

A number of groundwater monitoring studies have detected trichloroethylene in groundwater (see Section 5.4.2), which is further evidence of its leachability. The mobility of trichloroethylene in soil was demonstrated in a field study of river water infiltration to groundwater in which trichloroethylene was observed to leach rapidly into groundwater near sewage treatment plants in Switzerland (Schwarzenbach et al. 1983). No evidence of biological transformation of trichloroethylene in groundwater was found. Accurate prediction of trichloroethylene transport in groundwater is complicated by the sorption effect of organic and inorganic solids (Doust and Huang 1992).

Experimentally measured bioconcentration factors (BCFs), which provide an indication of the tendency of a chemical to partition to the fatty tissue of organisms, have been found to range between 10 and 100 for trichloroethylene in fish (Kawasaki 1980; Kenaga 1980; Neely et al. 1974; Veith et al. 1980). Barrows et al. (1980) estimated a value of 17 for bluegill sunfish. Somewhat lower BCFs were determined by Saisho et al. (1994) for blue mussel (4.52) and killifish (2.71). These numbers are suggestive of a low tendency to bioaccumulate.

Monitoring data on trichloroethylene concentrations in seawater and associated aquatic organisms are in agreement with the experimental BCF data. Concentrations of trichloroethylene (dry weight basis) detected in fish (eel, cod, coalfish, dogfish) from the relatively unpolluted Irish Sea ranged from below detection limits to 479 ppb (Dickson and Riley 1976). Levels of 2-56 ppb (wet weight) in liver tissue, and up to 11 ppb (wet weight) in other tissue, were found in various species of fish collected off the coast of Great Britain near several organochlorine plants (Pearson and McConnell 1975). Fish taken from the western coast of the United States near the discharge zone of the Los Angeles County waste-water treatment plant contained trichloroethylene levels of up to 6 ppb (wet weight) in liver

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tissue (Gossett et al. 1983). Clams and oysters from Lake Pontchartrain near New Orleans had trichloroethylene levels averaging between 0.8 and 5.7 ppb (wet weight) (Ferrario et al. 1985).

To assess bioaccumulation in the environment, the levels of trichloroethylene in the tissues of a wide range of organisms were determined (Pearson and McConnell 1975). Species were chosen to represent several trophic levels in the marine environment. The maximum overall increase in concentration between sea water and the tissues of animals at the top of food chains, such as fish liver, sea bird eggs, and sea seal blubber, was less than 100-fold for trichloroethylene. Biomagnification in the aquatic food chain does not appear to be important (Pearson and McConnell 1975).

Trichloroethylene has also been detected in small amounts in fruits and vegetables, suggesting a potential for bioconcentration in plants (see Section 5.4.4), although some of the trichloroethylene may have been a result of exposure after harvesting. Laboratory studies with carrot and radish plants and radioactively labelled trichloroethylene revealed that uptake occurred mainly through the foliage as opposed to the roots in these plants, although subsequent translocation resulted in substantial distribution throughout the plants (Schroll et al. 1994). The study authors determined fairly moderate BCFs of between 4.4 and 63.9.

### 5.3.2 Transformation and Degradation

#### 5.3.2.1 Air

The dominant transformation process for trichloroethylene in the atmosphere is reaction with photochemically produced hydroxyl radicals (Singh et al. 1982). Using the recommended rate constant for this reaction at 25°C ( $2.36 \times 10^{12} \text{ cm}^3 / \text{molecule-second}$ ) and a typical atmospheric hydroxyl radical concentration ( $5 \times 10^5 \text{ molecules/cm}^3$ ) (Atkinson 1985), the half-life can be estimated to be 6.8 days. Class and Ballschmiter (1986) state it as between 3 and 7 days. It should be noted that the half-lives determined by assuming first-order kinetics represent the calculated time for loss of the first 50% of trichloroethylene; the time required for the loss of the remaining 50% may be substantially longer.

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The reaction of volatile chlorinated hydrocarbons with hydroxyl radicals is temperature dependent and thus varies with the seasons, although such variation in the atmospheric concentration of trichloroethylene may be minimal because of its brief residence time (EPA 198%). The degradation products of this reaction include phosgene, dichloroacetyl chloride, and formyl chloride (Atkinson 1985; Gay et al. 1976; Kirchner et al. 1990). Reaction of trichloroethylene with ozone in the atmosphere is too slow to be an effective agent in trichloroethylene removal (Atkinson and Carter 1984).

### 5.3.2.2 Water

Oxidation of trichloroethylene in the aquatic environment does not appear to be a significant fate process, probably because of its having already been oxidized by the chlorine atoms. The rate of hydrolysis is also too slow to be an important transformation process (EPA 1979b). A study by Jensen and Rosenberg (1975) indicated that the rate of volatilization of trichloroethylene proceeds more rapidly than photooxidation or hydrolysis. Studies of photolysis and hydrolysis conducted by Chodola et al. (1989) demonstrated that photolysis did not contribute substantially to the transformation of trichloroethylene. Chemical hydrolysis appeared to occur only at elevated temperature in a high pH environment and, even then, at a very slow rate. Studies of the degradation of trichloroethylene in water during ultraviolet irradiation indicated that degradation decreased with increases in the total organic content of the water (Beltran et al. 1995).

Results from experiments conducted at high pH and temperature were extrapolated to pH 7 and 25°C (Jeffers et al. 1989), and the estimated half-life was  $1.3 \times 10^6$  years, which suggests that hydrolysis does not occur under normal environmental conditions. In contrast, estimates of the hydrolysis half-life of trichloroethylene under corresponding conditions were cited in other studies as about 10.7 months (Dilling et al. 1975) and 30 months (Pearson and McConnell 1975). It is not clear why there is such a large difference between these values; however, errors inherent in the extrapolation method used in the first approach (Jeffers et al. 1989) and the presence of transformation factors other than-chemical hydrolysis, such as microbial degradation, in the second approach (Dilling et al. 1975; Pearson and McConnell 1975) may account for the discrepancy in the numbers.

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An aerobic degradation study of trichloroethylene in seawater showed that 80% of trichloroethylene was degraded in 8 days (Jensen and Rosenberg 1975). Degradation products were not reported. Another study using domestic waste water as a microbial inoculum found that after the 1st week of incubation, 64% and 38% degradation was achieved for initial trichloroethylene concentrations of 5 and 10 ppm, respectively (Tabak et al. 1981). After the 4<sup>th</sup> week of incubation, these percentages were 87% and 84%, respectively. Microbial degradation products of trichloroethylene in groundwater were reported to be dichloroethylene and vinyl chloride (Smith and Dragun 1984).

Biotransformation was also strongly indicated as a factor in the degradation of trichloroethylene in a case of soil and groundwater pollution (Milde et al. 1988). The only ethylenes at the point source of pollution were tetrachloroethylene and trichloroethylene; however, substantial amounts of known metabolites of these two compounds (dichloroethylene, vinyl chloride, and ethylene) were found at points far from the source. Data from laboratory studies by the same group supported the study authors' contention that degradation was due to reductive dehalogenation by microorganisms. Microcosm studies of trichloroethylene biotransformation in aquifers have also indicated that reductive dehalogenation is the primary degradation reaction (Parsons et al. 1985; Wilson et al. 1986). However, a field study of groundwater at the Lawrence Livermore National Laboratory found a highly oxidized environment in which no evidence of reductive dehalogenation of trichloroethylene was seen (McNab and Narasimhan 1994).

Since neither biodegradation nor hydrolysis occurs at a rapid rate, most trichloroethylene present in surface waters can be expected to volatilize into the atmosphere. However, because trichloroethylene is denser than and only moderately soluble in water, that which is not immediately volatilized may be expected to submerge and thus be removed from contact with the surface (Doust and Huang 1992).

### 5.3.2.3 Sediment and Soil

The majority of trichloroethylene present on soil surfaces will volatilize to the atmosphere or leach into the subsurface. Once trichloroethylene leaches into the soil, it appears not to become chemically transformed or undergo covalent bonding with soil components. When trichloroethylene was absorbed onto kaolinite and bentonite, the <sup>13</sup>C nuclear magnetic resonance (CNMR) spectra showed no evidence of chemical reactions (Jurkiewicz and Maciel 1995). Because trichloroethylene is a dense nonaqueous

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phase liquid, it can move through the unsaturated zone into the saturated zone where it can displace soil pore water (Wershaw et al. 1994).

Biodegradation is favored only under limited conditions. When soil samples containing subsurface bacteria from depths of 1.2, 3.0, and 5.0 meters in a flood plain in Oklahoma were incubated with trichloroethylene for 16 weeks at 20°C, no detectable degradation of the chemical occurred (Wilson et al. 1983a). It has been shown that the biodegradation of trichloroethylene in soil increases with the organic content of the soil (Barrio-Lage et al. 1987). There is evidence that trichloroethylene may inhibit total soil biomass and fungi (Kanazawa and Filip 1986), possibly resulting in the inhibition of microbial transformation processes. However, the same authors observed an increase in anaerobic and specialized aerobic bacteria, which might indicate an opportunistic response to a suitable substrate by these microorganisms.

Degradation of trichloroethylene by anaerobes via reductive dehalogenation can be problematic because a common product is vinyl chloride, a known carcinogen (Ensley 1991). In an anaerobic column operated under methanogenic conditions, 100% transformation of injected tetrachloroethylene and trichloroethylene to vinyl chloride was obtained after 10 days (Vogel and McCarty 1985). Addition of electron donors was demonstrated to promote further degradation to the more benign compound ethylene (Freedman and Gossett 1989).

Anaerobic incubations of trichloroethylene with soils collected from lotus, rice, and vegetable fields in Japan resulted in biodegradation rates which varied with soil type, temperature, and initial concentration of trichloroethylene (Yagi et al. 1992). The lotus field soils degraded more than 80% of the trichloroethylene after 42 days, while the degradation in vegetable field soils was minimal. A study by Walton and Anderson (1990) compared soil samples collected from a former chlorinated solvent disposal site and microbial degradation of trichloroethylene in vegetated (grass, a legume, a composite herb) and nonvegetated soils. Biomass determinations, disappearance of trichloroethylene from the headspace of spiked soil slurries, and mineralization of  $^{14}\text{C}$ -trichloroethylene to radiolabelled carbon dioxide ( $^{14}\text{CO}_2$ ) all showed that microbial activity is greater in vegetated soils and that trichloroethylene degradation occurs faster in the vegetated than in the nonvegetated soils. An anaerobic bacterium that dechlorinates tetrachloroethylene and trichloroethylene to ethylene using

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hydrogen as the electron donor has been isolated (Maymo-Gate11 et al. 1997). The isolated strain did not appear to belong to any presently known genus or species.

Aerobic biodegradation of trichloroethylene occurs by cometabolism with aromatic compounds (Ensley 1991) and thus requires a cosubstrate such as phenol (Nelson et al. 1987, 1988) or toluene (Fan and Scow 1993). Trichloroethylene degradation by toluene-degrading bacteria has been demonstrated in the presence, but not absence, of toluene (Mu and Scow 1994). Isoprene, a structural analog of trichloroethylene, has also been used as a cosubstrate for trichloroethylene oxidation by some bacteria (Ewers et al. 1990). One source of inhibition of degradation in the absence of cosubstrate may be the toxicity of trichloroethylene itself to indigenous bacteria.

Bacteria have been found that use methane as an energy source and simultaneously degrade trichloroethylene using methane monooxygenase (Alvarez-Cohen and McCarty 1991a, 1991b; Bowman et al. 1993; Eng et al. 1991; Fox et al. 1990; Henry and Grbic-Galic 1991a, 1991b; Oldenhuis et al. 1991). Methane-utilizing bacteria were shown to aerobically degrade trichloroethylene to carbon dioxide in soil columns perfused with natural gas within 2 weeks (Wilson and Wilson 1985). Methanotrophs isolated from sediment likewise degraded 650 ng/mL of trichloroethylene in liquid culture to 200 ng/mL in 4 days (at 20°C), producing carbon dioxide and no dichloroethylene or vinyl chloride (Fogel et al. 1986). A possible reason for the persistence of trichloroethylene in the environment despite these natural decomposition processes lies in the sensitive balance which must be maintained between enough cosubstrate to induce the degrading enzymes and too much cosubstrate, which could outcompete the trichloroethylene and inhibit its decomposition (Ensley 1991). Such balance may rarely be achieved in nature.

## 5.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

### 5.4.1 Air

Monitoring data for trichloroethylene in ambient air in the United States, prior to 1981, were compiled by Brodzinsky and Singh (1982). This compilation, which includes over 2,300 monitoring points, reported mean trichloroethylene concentrations of 0.03 ppb in rural/remote areas, 0.460 ppb in urban/suburban areas, and 1.2 ppb in areas near emission sources of trichloroethylene. A similar

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compilation (EPA 1979a), which includes additional U.S. monitoring data and worldwide data, indicates that the ambient air mixing ratio of trichloroethylene is 0.01-0.03 ppb in the northern hemisphere and <0.003 ppb in the southern hemisphere. Slightly lower ambient air mixing ratios of 0.005-0.01 ppb have also been reported for the northern hemisphere (Class and Ballschmiter 1986; Fabian 1986).

Ambient air monitoring studies in the United States detected trichloroethylene concentrations of 0.24-3.9  $\mu\text{g}/\text{m}^3$  (0.04-0.72 ppb) in Portland, Oregon, in 1984 (Ligocki et al. 1985); 2.1  $\mu\text{g}/\text{m}^3$  (0.39 ppb) in Philadelphia, Pennsylvania, in 1983-1984 (Sullivan et al. 1985); 0.21-0.59 ppb in three New Jersey cities during the summer of 1981 and winter of 1982 (Harkov et al. 1984); and 0.09-0.225 ppb in seven cities (Houston, Texas; St. Louis, Missouri; Denver, Colorado; Riverside, California; Staten Island, New York; Pittsburgh, Pennsylvania; and Chicago, Illinois) in 1980-1981 (Singh et al. 1982). In these studies, levels were found to vary between the fall/winter season and the spring/summer season, with fall/winter levels usually higher. This is consistent with the observation that higher temperatures increase the rate of reaction with hydroxyl radicals and subsequent degradation of trichloroethylene (see Section 5.3.2.1).

Data gathered from several sites near Niigata, Japan, between April 1989 and March 1992 showed elevated levels of trichloroethylene and other volatile chlorinated hydrocarbons in the winter (Kawata and Fujieda 1993). A rural site in this study had annual mean concentrations between 0.17 and 0.32 ppb, while four industrial sites had mean concentrations between 0.029 and 4.8 ppb. The average trichloroethylene level detected in samples collected from ambient air in the Norwegian Arctic between 1982 and 1983 was 0.007 ppb (Hov et al. 1984). Average concentrations of trichloroethylene in Alaskan Arctic haze between 1980 and 1982 were 0.036 ppb in winter and 0.007 ppb in summer (Khalil and Rasmussen 1983).

Data collected from several locations in the city of Hamburg, Germany, showed ambient air concentrations of trichloroethylene ranging from 0.8 to 18.5  $\mu\text{g}/\text{m}^3$  (0.15 to 3.44 ppb) (Bruckmann et al. 1988). A monitoring study in Finland reported levels of 0.27 and 36  $\mu\text{g}/\text{m}^3$  (0.05 and 6.70 ppb) in ambient air from a suburban area and an industrialized area, respectively (Kroneld 1989). No trichloroethylene was detected in samples of rural air in that study.

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Some elevated outdoor air levels of trichloroethylene reported are associated with waste disposal sites. Average trichloroethylene levels of 0.08-2.43 ppb were detected in ambient air at six landfill sites in New Jersey; the maximum concentration was 12.3 ppb (Harkov et al. 1985). Levels between 3.0 and 3.2  $\mu\text{g} / \text{m}^3$  (0.56 ppb and 0.60 ppb) were found at a distance of 0.5-1.5 meters above the surface of a landfill known to contain halogenated volatile organic compounds in Germany (Koenig et al. 1987).

A survey of indoor air showed median concentrations of trichloroethylene as high as 27  $\mu\text{g} / \text{m}^3$  (5.0 ppb) in a North Carolina office building; 0.74  $\mu\text{g} / \text{m}^3$  (0.14 ppb) in a Washington, DC, school; and 0.82  $\mu\text{g} / \text{m}^3$  (0.15 ppb) in a Washington, DC, home for the elderly (Hartwell et al. 1985). The level of trichloroethylene in the air of an indoor university laboratory was 0.008 ppm (8.0 ppb) (Nicoara et al. 1994). Based on the properties of trichloroethylene and a three-compartment model, the levels of trichloroethylene in indoor air have been estimated (McKone 1987). If the tap water contained 1 mg/L, the air in the shower during use was estimated to be 3.3 ppm, while in the rest of the house it was estimated to be 0.02 ppm during the day (7 am-1 pm) and 0.0045 ppm during the night (11 pm-7 am).

### 5.4.2 Water

The concentration of trichloroethylene in the open oceans may be an indication of the environmental background levels in water. Levels in open waters of the Gulf of Mexico were below the detection level of 1 ppt (Sauer 1981). Average levels of 7 ng/L (7 ppt) and 0.3 ppt were found in the northeastern Atlantic (Murray and Riley 1973) and in Liverpool Bay (Pearson and McConnell 1975), respectively.

Rain water collected in Portland, Oregon, in 1984 contained trichloroethylene levels of 0.78-16 ng/L (0.78-16 ppt) (Ligocki et al. 1985). An average trichloroethylene concentration of 5 ng/L (5 ppt) was found in rain water from La Jolla, California, and levels of 30 and 39 ppt were identified in snow from southern California and Alaska, respectively (Su and Goldberg 1976). Levels up to 150 ng/L (150 ppt) were found in samples collected in rainwater in industrial cities in England (Pearson and McConnell 1975). Rainwater samples collected in Tokyo between October 1989 and September 1990 had a mean trichloroethylene level of 136 ng/L (136 ppt), with higher levels in samples obtained during the winter (Jung et al. 1992).



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Trichloroethylene has been detected in many samples taken from drinking water supplied by contaminated sources from which trichloroethylene and other volatile organic compounds are not always completely removed by conventional water treatment. The EPA Groundwater Supply Survey of finished water from 945 drinking water systems nationwide using groundwater sources found trichloroethylene in 91 water systems (detection limit 0.2 ppb); the median level of the positive samples was approximately 1 µg /L (1 ppb), with a single maximum level of 130 µg /L (130 ppb) (Westrick et al. 1984). Trichloroethylene levels ranging from 10 to 250 ng/L (0.01-0.25 ppb) were found in tap water from homes in the vicinity of the Love Canal waste site in New York (Barkley et al. 1980). In other countries, thirty Canadian drinking water sources were found to contain trichloroethylene levels ranging from <1 to 2 ppb (Otson et al. 1982), and recent drinking water samples from Zagreb, Croatia, contained 0.69-35.9 µg /L (0.69-35.9 ppb) (Skender et al. 1993).

A summary of U.S. groundwater analyses from both federal and state studies reported that trichloroethylene was the most frequently detected organic solvent and the one present in the highest concentration (Dyksen and Hess 1982). Trichloroethylene was detected in 388 of 669 groundwater samples collected in New Jersey from 1977 to 1979, with a maximum concentration of 635 ppb (Page 1981). Maximum concentrations ranging from 900 to 27,300 ppb trichloroethylene were found in contaminated wells from four states (Pennsylvania, New York, Massachusetts, and New Jersey) (Burmester 1982).

A possible source for much of the groundwater contamination is landfill leachate containing trichloroethylene. Trichloroethylene was the most commonly found chemical at NPL sites in New York State (Mumtaz et al. 1994). The compound was detected in leachate samples from Minnesota municipal solid waste landfills at levels ranging from 0.7 to 125 µg /L (0.7-125 ppb) and in groundwater near landfills at levels ranging from 0.2 to 144 µg /L (0.2-144 ppb) (Sabel and Clark 1984). Trichloroethylene was also detected in landfill leachate from a landfill in New Jersey at concentrations of up to 7,700 µg /L (7,700 ppb) (Kosson et al. 1985). Trichloroethylene has also been detected in ground water at the U.S. Army Cold Regions Research and Engineering Laboratory in Hanover, NH, where it was used as a refrigerant between 1960 and 1987 (Hewitt and Shoop 1994). In water collected directly after well instillation, the trichloroethylene concentrations were 0.044-180 ppm.

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An analysis of the EPA E'TORET Data Base (1980-1982) found that trichloroethylene had been positively detected in 28% of 9,295 surface water reporting stations nationwide (Staples et al. 1985). An analysis of 1,350 samples taken from 1978 to 1979 and 4,972 samples from 1980 to 1981 from the Ohio River system found a similar percentage of positive detections; most positive samples had trichloroethylene levels of 0.1-1.0 ppb (Ohio River Valley Sanitation Commission 1980, 1982). Trichloroethylene was detected in 261 of 462 surface water samples collected in New Jersey from 1977 to 1979, with a maximum concentration of 32.6 ppb (Page 1981). Mean levels of 0.008-0.13 µg /L (0.008-0.13 ppb) trichloroethylene were found in the Niagara River and Lake Ontario in 1981 (Strachan and Edwards 1984).

### 5.4.3 Sediment and Soil

A maximum trichloroethylene level of 9.9 ppb was found in sediment from Liverpool Bay, England (Pearson and McConnell 1975). Sediment levels from nondetectable to 0.2 ppb (wet weight) trichloroethylene were found in Lake Pontchartrain near New Orleans (Ferrario et al. 1985). An analysis of the EPA STORET Data Base (1980-1982) found that trichloroethylene had been positively detected in sediment samples taken at 6% of 338 observation stations, with median levels of <5 µg /kg (dry weight) (<5 ppb) (Staples et al. 1985). The observation stations included both "ambient" and "pipe" sites. Ambient sites include streams, lakes, and ponds and are intended to be indicative of general U.S. waterway conditions. Pipe sites refer to municipal or industrial influents or effluents.

Trichloroethylene was qualitatively detected in the soil/sediment matrix of the Love Canal waste site near Niagara Falls (Hauser and Bromberg 1982). Sediment concentrations were found to be <0.5 µg /kg (dry weight) (<0.5 ppb) near a discharge point for effluent containing 17 ppb trichloroethylene in Los Angeles (Gossett et al. 1983).

Trichloroethylene in soil and groundwater were found to be correlated ( $r^2$  0.9994) in samples taken during well instillation at the U.S. Army Cold Regions Research and Engineering Laboratory in Hanover, NH (Hewitt and Shoop 1994). Concentrations of trichloroethylene in soil from the saturated zone were 0.008-25 mg/kg, while concentrations in the groundwater were 0.044-180 ppm.

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### 5.4.4 Other Environmental Media

Trichloroethylene has been detected in dairy products (milk, cheese, butter) at 0.3-10  $\mu\text{g}/\text{kg}$  (0.3-10 ppb), meat (English beef) at 12-16 ppb, oils and fats at 0-19 ppb, beverages (canned fruit drink, light ale, instant coffee, tea, wine) at 0.02-60 ppb, fruits and vegetables (potatoes, apples, pears, tomatoes) at 0-5 ppb, and fresh bread at 7 ppb (McConnell et al. 1975). Samples obtained from a food processor in Pennsylvania contained trichloroethylene concentrations of 68 ppb in plant tap water, 28 ppb in Chinese-style sauce, 40 ppb in quince jelly, 25 ppb in crab apple jelly, 20 ppb in grape jelly, and 50 ppb in chocolate sauce (Entz and Hollifield 1982). Various samples of U.S. margarine were found to contain trichloroethylene levels of 440-3,600  $\text{ng}/\text{g}$  (440-3,600 ppb) (Entz et al. 1982). An analysis of intermediate grain-based foods in 1985 found the following trichloroethylene levels (in ppb concentrations): corn muffin mix (0.0); yellow corn meal (2.7); fudge brownie mix (2.4); dried lima beans (0.0); lasagna noodles (0.0); bleached flour (0.77); uncooked rice (0.0); and yellow cake mix (1.3) (Heikes and Hopper 1986).

Another study found that trichloroethylene can be absorbed from the atmosphere by foods and concentrated over time, so that acceptable ambient air levels may still result in food levels which exceed acceptable limits (Grob et al. 1990). The authors estimated that in order to limit food concentrations of trichloroethylene to 50  $\mu\text{g}/\text{kg}$  (the maximum tolerated limit for food halocarbons in Switzerland), the level in surrounding air should not exceed 38.5  $\mu\text{g}/\text{m}^3$  (0.007 ppm). Since the accepted levels found near emission sources are often far above this limit, foods processed or sold near these sources may routinely exceed the tolerated trichloroethylene concentration, thus making the setting of air emission standards problematic. It is also noteworthy that the limits recommended by Grob et al. (1990) exceed acceptable ambient air concentrations for many regions of the United States (see Chapter 7).

An analysis of six municipal solid waste samples from Hamburg, Germany, revealed levels of trichloroethylene ranging from undetectable to 0.59  $\text{mg}/\text{kg}$  (Deipser and Stegmann 1994). In a study analyzing automobile exhaust for chlorinated compounds, trichloroethylene was not detected (Hasanen et al. 1979).

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### 5.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

The most important routes of exposure to trichloroethylene for most members of the general population appear to be inhalation of the compound in ambient air and ingestion of drinking water. Available data indicate that dermal exposure is not an important route for most people. General population exposure from inhalation of ambient air varies widely depending on location. In general, rural areas exhibit lower background concentrations of trichloroethylene as compared to urban areas. One study comparing differences in trichloroethylene levels reported a significant difference in values between rural and urban workers with average blood trichloroethylene levels of 0.180 ng/L and 0.763 ng/L, respectively (Brugnone et al. 1994). A study of an urban population was conducted using the residents of the city of Zagreb, Croatia (Skender et al. 1994). Blood concentration levels of trichloroethylene and tetrachloroethylene among the residents ranged from <0.015 to 0.090 µg /L. The concentrations in drinking water in the city ranged from <0.05 to 22.93 mg/L and from 0.21 to 7.80 µg /L for trichloroethylene and tetrachloroethylene, respectively.

Assuming a typical air concentration range of 100-500 ppt (Singh et al. 1981, 1982) and a breathing rate of 20 m<sup>3</sup> air/day, the average daily air intake of trichloroethylene can be estimated at 11-33 mg/day. Average daily water intake of trichloroethylene can be estimated at 2-20 mg/day, assuming a typical concentration range of 2-7 ppb and consumption of 2 L water/day.

Because of the high propensity of trichloroethylene to volatilize from water, inhalation may be a major route of exposure in homes supplied with contaminated water (Andelman 1985b). In two homes (using well water containing the relatively high level of 40,000 ppb trichloroethylene), a running shower was found to elevate trichloroethylene levels in bathroom air from <0.5 to 81 mg/ m<sup>3</sup> (93 to 15,072 ppb) in less than 30 minutes (Andelman 1985a). Significantly elevated indoor air levels of trichloroethylene (as compared to normal outdoor levels) have been found in various buildings, but the elevated levels seem to be related to new building construction using products containing trichloroethylene solvents or consumer products containing trichloroethylene (Hartwell et al. 1985; Wallace et al. 1987).

Trichloroethylene levels monitored in expired breath of 190 New Jersey residents were correlated with personal exposure levels, which were consistently higher than outdoor air levels and were instead

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attributed to indoor air levels (Wallace et al. 1985). Other studies have expanded upon and confirmed these findings, concluding that indoor air is a more significant exposure source of trichloroethylene than outdoor air, even near major point sources such as chemical plants (Wallace 1986; Wallace et al. 1986a, 1986b, 1986c, 1986d). Wallace et al. (1989) reported air concentrations for four homes (nine samples per home) in North Carolina and found that indoor air concentrations of trichloroethylene in all homes were consistently higher than the outdoor concentrations. In fact, trichloroethylene did not have a measurable median outdoor air concentration, while median indoor values ranged from 0.95 to 26  $\mu\text{g} / \text{m}^3$  (0.2-4.8 ppb).

Correlations of exposure with other measures of body burden are often difficult and their results are consequently less conclusive. For example, trichloroethylene was present at unspecified levels in eight of eight samples of mother's milk from four urban areas in the United States (Pellizzari et al. 1982). Whole-blood specimens from 121 men and 129 women with no known exposure to trichloroethylene had levels from nondetectable to 1.5 ppb (Antoine et al. 1986). Post-mortem analyses of human tissue revealed body fat levels of 1.4-32  $\mu\text{g} / \text{kg}$  (1.4-32 ppb) (wet weight) among males and females with unspecified exposures (McConnell et al. 1975).

Various consumer products have been found to contain trichloroethylene. These include wood stains, varnishes, and finishes; lubricants; adhesives; typewriter correction fluids; paint removers; and cleaners (Frankenberry et al. 1987). Trichloroethylene use as an inhalation anesthetic, fumigant, and extractant for decaffeinating coffee has been discontinued in the United States (EPA 1985c).

Contamination of drinking water supplies with trichloroethylene varies with location and with the drinking water source (surface water or groundwater). Generally higher levels are expected in groundwater because trichloroethylene volatilizes rapidly from surface water. There is some evidence that trichloroethylene can be produced in small amounts during the chlorination process of waste-water treatment (Bellar et al. 1974), although no evidence exists for its formation through drinking water chlorination (Westrick et al. 1984).

The National Occupational Exposure Survey (NOES), conducted by NIOSH from 1981 to 1983, estimated that 401,000 workers employed at 23,225 plant sites were potentially exposed to trichloroethylene in the United States (NOES 1990). The NOES database does not contain

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information on the frequency, concentration, or duration of exposures; the survey provides only estimates of workers potentially exposed to chemicals in the workplace.

The majority of data regarding worker exposure to trichloroethylene was obtained from degreasing operations, which is the primary industrial use of trichloroethylene. Worker exposure data indicated that exposure is likely to vary, although mean TWA concentrations were generally consistent and usually ranged from  $\leq 50$  to 100 ppm (Santodonato 1985). OSHA allows an 8-hour TWA permissible exposure limit of 100 ppm. The 15-minute TWA exposure, which should not be exceeded at any time during a workday, is 300 ppm (OSHA 1993). Higher than normal workplace exposure was generally attributable to poor workplace practices (improper operating procedures, negligence with regard to equipment maintenance or repair) and/or inadequate engineering controls. TWA concentrations from personal monitoring ranged from 1.2 to 5.1 ppm at individual industrial sites where trichloroethylene was used during the process of filling spray cans with insecticide and where trichloroethylene was used as a solvent during the formation of fiberglass aircraft components (Santodonato 1985).

### 5.6 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Because of the pervasiveness of trichloroethylene in the environment, most people are exposed to it through drinking water, air, or food, although the levels of exposure are probably far below those causing any adverse effects. Concern may be justified, however, for people who are continuously exposed to elevated levels, such as residents of some urban or industrialized areas, people living near waste facilities, or people exposed at work. Short-term exposure to high levels of trichloroethylene may also pose risks for people using products containing the chemical in areas with inadequate ventilation. The discontinuation of trichloroethylene use in many medical applications and some consumer products has generally decreased the exposure risks in these situations.

As a result of volatilization, significantly elevated indoor air levels of trichloroethylene can occur in homes that use water supplies contaminated with trichloroethylene (Andelman 1985a). The transfer of trichloroethylene from shower water to air in one study had a mean efficiency of 61% which was independent of water temperature (McKone and Knezovich 1991). The study authors concluded that showering for 10 minutes in water contaminated with trichloroethylene could result in a daily exposure by inhalation comparable to that expected by drinking contaminated tap water. Another study using a

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model shower system found that, in addition to shower spray, shower water collecting around the drain could be an important source of volatilized trichloroethylene, and the fraction volatilized could be affected by spray drop size and flow rate (Giardino et al. 1992).

A survey of 20 brands of typographical correction fluids found that several contained 10% or less trichloroethylene, although other volatile organic compounds present at higher levels probably posed a greater hazard to people using these products (Ong et al. 1993). Various other consumer products have been found to contain trichloroethylene, such as paint removers, strippers, adhesives, and lubricants (Frankenberry et al. 1987).

Workers involved in the manufacture or use of trichloroethylene as a metal degreaser or general solvent may constitute a group at risk because of the potential for occupational exposure. Occupational exposure to trichloroethylene may also occur during its use as a chemical intermediate in the production of polyvinyl chloride (McNeill 1979).

An EPA TEAM (Total Exposure Assessment Methodology) study conducted in New Jersey attempted to identify factors associated with risk of higher inhalation of trichloroethylene (Wallace et al. 1986b). The following factors (in order of importance) were identified: wood processing, working at a plastics plant, exposure to a gas furnace, working at a scientific lab, and smoking.

### 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 trichloroethylene 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 trichloroethylene.

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

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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.** The physical and chemical properties of trichloroethylene are well characterized (HSDB 1994; McNeill 1979; Windholz 1983) and allow prediction of the environmental fate of the compound. Estimates based on available constants are generally in good agreement with experimentally determined values. No additional studies are required at this time.

**Production, Import/Export, Use, Release, and Disposal.** Humans are at risk of exposure to trichloroethylene because of its widespread use and distribution in the environment. Production, import, and use of the chemical are known to be relatively high, but recent quantitative data were not available (HSDB 1994). Trichloroethylene is released to the atmosphere mainly through its use in vapor degreasing operations (EPA 1985e). Landfills can be a concentrated source of trichloroethylene on a local scale. It is also released to surface water and land in sewage sludges and industrial liquid or solid waste. Trichloroethylene is considered a hazardous waste and its disposal is subject to regulations (see Chapter 7). More current data on production, use in food processing and consumer products, releases, efficiency of disposal practices, adequacy of current disposal regulations, and the extent of recovery and recycling of trichloroethylene would assist in estimating human potential exposures, particularly of populations living near industrial facilities and hazardous waste sites.

According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit chemical release and off-site transfer information to the EPA. The Toxics Release Inventory (TRI), which contains this information for 1992, became available in 1994. This database will be updated yearly and should provide a list of industrial production facilities and emissions.

**Environmental Fate.** Trichloroethylene released to environment partitions mainly to the atmosphere (EPA 1985e). The compound is transported in atmosphere, groundwater, and soil. Trichloroethylene is transformed in the atmosphere by photooxidation (Singh et al. 1982). Trichloroethylene is expected to volatilize very rapidly from surface water and soil (EPA 1985c; Park et al. 1988). Trichloro-



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ethylene is biodegraded in water (Jensen and Rosenberg 1975; Smith and Dragun 1984) and, to a limited extent, in soil (Maymo-Gate11 et al. 1997; Yagi et al. 1992). Trichloroethylene may persist in groundwater. Additional information on the anaerobic degradation of trichloroethylene in groundwater and on the rates of transformation in soil is needed to define the relative importance of these media as potential pathways for human exposure.

**Bioavailability from Environmental Media.** Trichloroethylene can be absorbed following inhalation (Andersen et al. 1980; Astrand and Ovrum 1976; Bartonicek 1962; Dallas et al. 1991; Fernandez et al. 1977; Monster et al. 1976; Mtiller et al. 1974; Sato and Nakajima 1978), oral (DeFalque 1961; D'Souza et al. 1985; Klienfeld and Tabershaw 1954; Prout et al. 1985; Stephens 1945; Stevens et al. 1992; Templin et al. 1993; Withey et al. 1983), or dermal (Bogen et al. 1992; Jakobson et al. 1982; McCormick and Abdul-Rahman 1991; Sato and Nakajima 1978; Steward and Dodd 1964; Tsuruta 1978) exposure. All these routes of exposure may be of concern to humans because of the potential for trichloroethylene to contaminate the air, drinking water, food, and soil. More information on the absorption of trichloroethylene following ingestion of contaminated soil and plants grown in contaminated soil near hazardous waste sites are needed to determine bioavailability of the compound in these media.

**Food Chain Bioaccumulation.** Information is available regarding bioaccumulation potential in aquatic food chains. Studies show that trichloroethylene has a low-to-moderate bioconcentration potential in aquatic organisms (Pearson and McConnell 1975) and some plants (Schroll et al. 1994). Information is needed, however, regarding bioaccumulation potential in terrestrial food chains.

**Exposure Levels in Environmental Media.** Reliable monitoring data for the levels of trichloroethylene in contaminated media at hazardous waste sites are needed so that the information obtained on levels of trichloroethylene in the environment can be used in combination with the known body burden of trichloroethylene to assess the potential risk of adverse health effects in populations living in the vicinity of hazardous waste sites. Trichloroethylene is widely distributed in the environment and has been detected in air (Brodzinsky and Singh 1982; Bruckmann et al. 1988; Class and Ballschmiter 1986; Fabian 1986; Harkov et al. 1985; Hartwell et al. 1985; Hov et al. 1984; Kawata and Fujieda 1993; Ligocki et al. 1985; Sullivan et al. 1985), water (Barkley et al. 1980; Burmaster 1982; Ligocki et al. 1985; Mumtaz et al. 1994; Murray and Riley 1973; Otson et al. 1982;

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Sauer 1981), soil (Hewitt and Shoop 1994; Hunter and Bromberg 1982), and food (Entz and Hollitield 1982; Entz et al. 1982; Grob et al. 1990; Heikes and Hopper 1986; McConnell et al. 1975). The levels of trichloroethylene in air, water, sediment, and foods are well documented, but some of these studies are not current. More recent data characterizing the concentration of trichloroethylene in drinking water, soils, and air surrounding hazardous waste sites and on estimates of human intake from these media are needed to assess human exposure to trichloroethylene for populations living near hazardous waste sites.

**Exposure Levels in Humans.** This information is necessary for assessing the need to conduct health studies on these populations. Trichloroethylene has been detected in human body fluids such as blood (Brugnone et al. 1994; Skender et al. 1994) and breast milk (Pellizzari et al. 1982). Most of the monitoring data have come from occupational studies of specific worker populations exposed to trichloroethylene. More information on exposure levels for populations living in the vicinity of hazardous waste sites is needed for estimating human exposure.

**Exposure Registries.** A subregistry has been established for trichloroethylene as part of the National Exposure Registry. There are 4,280 persons (along with general health data) enrolled on the subregistry (ATSDR 1994; Burg et al. 1995). This data is part of the public-user data files established and maintained by the Exposure and Disease Registry Branch, Division of Health Studies, ATSDR. The information that is amassed in the National Exposure Registry will facilitate the epidemiological research needed to assess adverse health outcomes that may be related to the exposure to this compound.

### 5.7.2 On-going Studies

As part of the Third National Health and Nutrition Evaluation Survey (NHANES III), the Environmental Health Laboratory Sciences Division of the National Center for Environmental Health, Centers for Disease Control, will be analyzing human blood samples for trichloroethylene and other volatile organic compounds. These data will give an indication of the frequency of occurrence and background levels of these compounds in the general population.

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Research on human exposure also includes studies by Dr. J.W. Gillett at Cornell University which model dermal absorption of trichloroethylene and other volatile organic compounds in a simulated shower and bath chamber. The development of a three-dimensional mathematical model for describing exposure from contaminated groundwater of residents living near superfund sites is the focus of work carried out by Dr. C.P. Weisel at Rutgers University. Measurements from a pilot scale soil system are included in the model, along with actual body burden measurements taken from residents, to assess the exposure risk from the transport of trichloroethylene in groundwater through soil and into the basements of homes. Health effects caused by ingestion of foods with fumigant residues, such as trichloroethylene, are being studied in rats by Dr. T. Shibamoto at the University of California, Davis, with the support of the (U.S. Department of Agriculture) USDA.

Several on-going projects are investigating the use of biotechnology to remediate sites contaminated with trichloroethylene. Dr. AK. Shiemke of West Virginia University is working on the isolation of a membrane-bound methane monooxygenase from trichloroethylene-degrading bacteria in order to study its mechanism of action, and related work is being carried out at Iowa State University by Dr. A.A. Dispirito. Reductive dechlorination of chlorinated organic compounds by anaerobes is being studied by Dr. S.H. Zinder at Cornell University. Continuing research on trichloroethylene biodegradation in soil columns under various conditions, including the presence of different cosubstrates and bacterial cultures, is being performed by Dr. K.M. Scow at the University of California, Davis. The potential of a system using methane application to stimulate the *in situ* biodegradation of trichloroethylene in groundwater by methanotrophs is being investigated by Dr. W.J. Jewell of Cornell University. A similar approach exploiting the nonspecific oxygenase activity of some nitrifying bacteria induced by ammonia and other water-soluble compounds is being studied by Dr. K.A. Sandbeck at Geomicrobial Technologies, Inc. Dr. H. Bohn at the University of Arizona is conducting USDA-supported research to discover the optimal redox conditions for trichloroethylene degradation in soil.

Nonbiological methods for removal of trichloroethylene from water are also being studied. These include the use of a hollow fiber membrane contactor (Dr. AK. Zander, Clarkson University), photocatalysis by solar or artificially irradiated semiconductor powders (Dr. G. Cooper, Photocatalytics, Inc.), and micellar-enhanced ultrafiltration (Dr. B.I. Roberts, Surfactant Associates, Inc.).

