6. POTENTIAL FOR HUMAN EXPOSURE

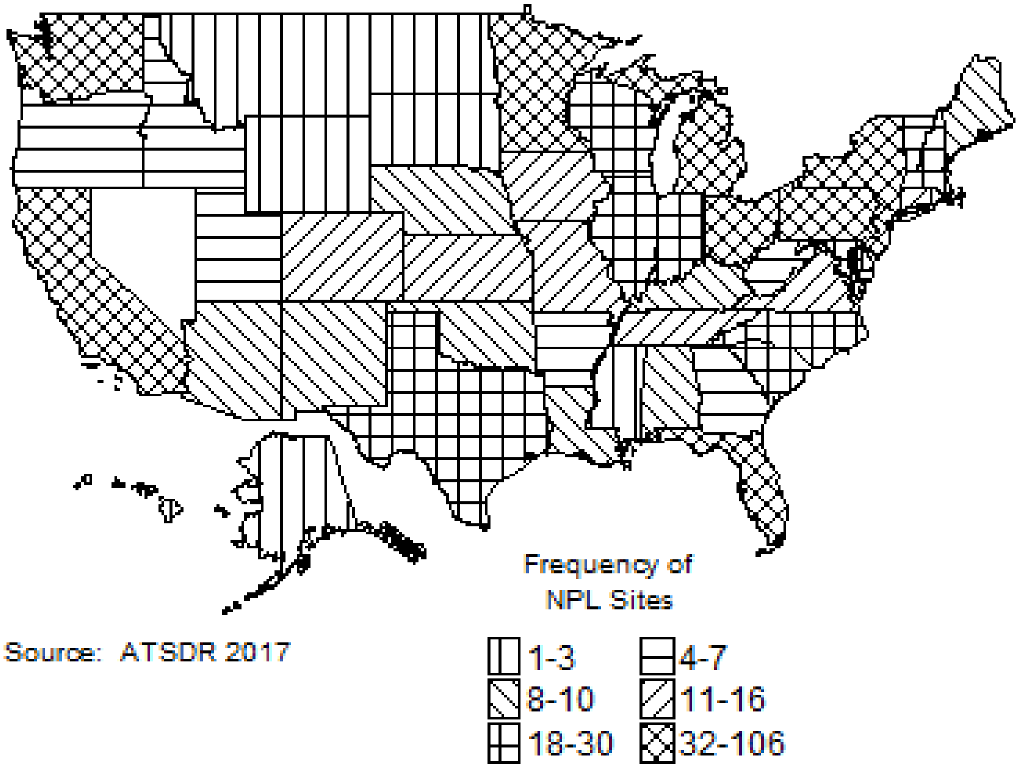
6.1 OVERVIEW

Trichloroethylene has been identified in at least 1,051 of the 1,854 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2017a). However, the number of sites evaluated for trichloroethylene is not known. The frequency of these sites can be seen in Figure 6-1. Of these sites, 1,043 are located within the United States, 1 is located in Guam, 1 is located in the Virgin Islands, and 6 are located in Puerto Rico (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. Trichloroethylene and other volatile organic chemicals may off-gas from contaminated groundwater and soil and migrate into air spaces beneath buildings to enter the indoor air, a process termed vapor intrusion (EPA 2002).

In general, atmospheric levels are highest in areas of concentrated industry and population and lower in rural and remote regions. Mean trichloroethylene concentrations measured in air at locations across the United States are generally between 0.01 and 0.3 ppb, although mean levels as high as 3.4 ppb have been reported (EPA 2011f). 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. Between 4.5 and 18% of the drinking water supply sources in the United States that are tested on a yearly basis by the U.S. EPA have some trichloroethylene contamination. Levels are typically <30 ppb. Trichloroethylene levels in the low ppb range have been measured in food; however, levels as high as 140 ppb were measured in a few samples. It should be noted that the amount of trichloroethylene found by chemical analysis is not necessarily the amount that is bioavailable.

Figure 6-1. Frequency of NPL Sites with Trichloroethylene Contamination



Source: ATSDR 2017

6.2 RELEASES TO THE ENVIRONMENT

The TRI is an annual compilation of information on the release of toxic chemicals by manufacturing and processing facilities. TRI data should be used with caution because only certain types of facilities are required to report (EPA 2005b). This is not an exhaustive list. Manufacturing and processing facilities are required to report information to the TRI only if they employ 10 or more full-time employees; if their facility is included in Standard Industrial Classification (SIC) Codes 10 (except 1011, 1081, and 1094), 12 (except 1241), 20–39, 4911 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4931 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4939 (limited to facilities that combust coal and/or oil for the purpose of generating electricity for distribution in commerce), 4953 (limited to facilities regulated under RCRA Subtitle C, 42 U.S.C. section 6921 et seq.), 5169, 5171, and 7389 (limited S.C. section 6921 et seq.), 5169, 5171, and 7389 (limited to facilities primarily engaged in solvents recovery services on a contract or fee basis); and if their facility produces, imports, or processes ≥25,000 pounds of any TRI chemical or otherwise uses >10,000 pounds of a TRI chemical in a calendar year (EPA 2005b).

6.2.1 Air

Estimated releases of 1,886,809 pounds (855.8 metric tons) of trichloroethylene to the atmosphere from 154 domestic manufacturing and processing facilities in 2017, accounted for about 92% of the estimated total environmental releases from facilities required to report to the TRI (TRI17 2018). These releases are summarized in Table 6-1.

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. Millet et al. (2009) estimated a trichloroethylene emission rate of 7.6 Gg/year (7,600 metric tons/year) based on aircraft measurements collected over the United States from 2004 to 2006.

Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Trichloroethylene[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
AL	3	99,952	0	0	0	780	99,952	780	100,732
AR	2	59	0	0	0	11	59	11	70
AZ	2	10,483	0	0	250	0	10,483	250	10,733
CA	2	45	0	0	0	0	45	No data	45
CO	1	6,560	0	0	0	0	6,560	No data	6,560
CT	4	56,090	0	0	0	0	56,090	No data	56,090
DE	1	1,275	0	0	0	0	1,275	No data	1,275
GA	8	17,005	0	0	0	0	17,005	No data	17,005
IA	1	10,306	0	0	0	0	10,306	No data	10,306
IL	15	98,618	0	0	5	2,013	98,618	2,018	100,636
IN	4	716,301	5	3,349	715	0	716,306	4,064	720,370
KS	6	106,509	1	374	306	116	106,884	422	107,306
KY	6	31,718	26	0	0	0	31,744	No data	31,744
LA	14	50,948	2	0	11	1,028	50,950	1,039	51,989
MA	5	28,747	0	0	0	1,436	28,747	1,436	30,183
ME	1	3,413	0	0	0	2,058	3,413	2,058	5,471
MI	2	11,581	0	0	0	0	11,581	No data	11,581
MN	7	81,337	0	0	0	0	81,337	No data	81,337
MS	8	39,902	0	0	0	0	39,902	No data	39,902
NE	2	28,119	0	0	632	0	28,119	632	28,751
NJ	2	8,898	0	0	0	0	8,898	No data	8,898
NM	1	118	0	0	90,876	0	90,994	No data	90,994
NV	1	500	0	0	0	0	500	No data	500
NY	3	7,498	0	0	0	0	7,498	No data	7,498
OH	13	101,363	0	0	0	27,570	101,363	27,571	128,934
OK	1	426	0	0	0	0	426	No data	426
OR	2	56,282	0	0	22,917	0	79,199	No data	79,199
PA	9	168,860	0	0	31	77	168,860	109	168,968
SC	4	5,444	0	0	0	0	5,444	No data	5,444
TN	3	113,282	0	0	0	0	113,282	No data	113,282
TX	10	4,444	0	0	38	324	4,444	362	4,806
UT	3	8,421	0	0	11	0	8,421	11	8,432
VA	2	1,962	0	0	0	0	1,962	No data	1,962
VT	1	11	0	0	0	0	11	No data	11

Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Trichloroethylene[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
WA	1	1,980	0	0	0	0	1,980	No data	1,980
WI	3	8,353	0	0	0	0	8,353	No data	8,353
Total	154	1,886,809	34	3,723	115,793	35415	2,001,010	40,764	2,041,774

[a] The TRI data should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list. Data are rounded to nearest whole number.
[b] Data in TRI are maximum amounts released by each facility.
[c] Post office state abbreviations are used.
[d] Number of reporting facilities.
[e] The sum of fugitive and point source releases are included in releases to air by a given facility.
[f] Surface water discharges, waste water treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).
[g] Class I wells, Class II-V wells, and underground injection.
[h] Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other onsite landfills, land treatment, surface impoundments, other land disposal, other landfills.
[i] Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown.
[j] The sum of all releases of the chemical to air, land, water, and underground injection wells.
[k] Total amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI17 2018 (Data are from 2017)

More recent data continue to suggest that degreasing operations represent the greatest emission source of trichloroethylene to the atmosphere; however, levels have been significantly reduced in the past 2 decades as new engineering controls and design features in degreasing operations have been initiated under the National Emissions Standard for Hazardous Air Pollutants (NESHAP), which limits emissions from degreasers using traditional solvents like trichloroethylene. Section 112(b) of the Clean Air Act (CAA) lists trichloroethylene as one of 188 hazardous air pollutants (HAPs) known to cause or suspected of causing cancer or other serious human health effects or ecosystem damage (EPA 2000). EPA's National Emission Inventory (NEI) database contains data regarding sources that emit criteria air pollutants and their precursors, and HAPs for the 50 United States, Washington DC, Puerto Rico, and the U.S. Virgin Islands (prior to 1999, criteria pollutant emission estimates were maintained in the National Emission Trends [NET] database and HAP emission estimates were maintained in the National Toxics Inventory [NTI] database). The NEI database derives emission data from multiple sources, including state and local environmental agencies; the TRI database; computer models for on- and off-road emissions; and databases related to EPA's Maximum Achievable Control Technology (MACT) programs to reduce emissions of HAPs. Using composite data from the NTI database from 1990 to 1993, it was estimated that the annual emissions of trichloroethylene in the United States was approximately 71,999 tons per year during that time frame (EPA 2000). More recent data illustrate the decline in trichloroethylene emissions in the United States. Data downloaded from the 2005 NEI (see Table 6-2) indicated that the total emission of trichloroethylene was approximately 9,911 tons, with the largest contribution arising from non-point solvent degreasing sources (EPA 2011a). Data downloaded from the 2008 NEI (see Table 6-3) indicated that the total emission of trichloroethylene decreased to 3,692 tons, with approximately 75% of the total arising from solvent degreasing (EPA 2013b). Data from the 2011 NEI (see Table 6-4) showed a further decline in trichloroethylene emissions to approximately 3,250 tons (EPA 2015b).

A 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 3,400 ng/g fresh weight/hour, with the higher rates seen in subtropical *Rhodophyta* species. The importance of this source of trichloroethylene could not be estimated 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. It is not known whether this source could be a potential concern as a major source of atmospheric emissions of trichloroethylene in coastal areas since there are many unknown factors. However, modeling of the distribution of

Table 6-2. 2005 NEI Total National Emissions

Source	Name	Annual emissions in tons
Nonpoint	Solvent; NEC	2,062.42
Nonpoint	Graphic arts	0.26
Nonpoint	Industrial process; NEC	5.43
Nonpoint	Industrial process; chemical manufacturing	0.10
Nonpoint	Fuel comb; commercial/institutional	0.01
Nonpoint	Indus process; metals	0.48
Nonpoint	Dry cleaning	346.96
Nonpoint	Solvent; non-industrial	241.64
Nonpoint	Surface coating; industrial	2.72
Nonpoint	Waste disposal	142.32
Nonpoint	Degreasing	2,645.34
Point	Degreasing	2,057.58
Point	Fuel comb; commercial/institutional	0.32
Point	Fuel comb; electric utility	0.71
Point	Fuel comb; industrial boilers, ICEs	1.35
Point	Graphic arts	4.99
Point	Dry cleaning	0.05
Point	Indus process; cement manufacturing	0.03
Point	Bulk gasoline terminals	0.00
Point	Indus process; metals	7.17
Point	Waste disposal; open burning	0.14
Point	Indus process; NEC	1,417.97
Point	Indus process; petroleum refineries	1.02
Point	Indus process; pulp and paper	37.71
Point	Indus process; storage and transfer	14.69
Point	Solvent; NEC	107.15
Point	Surface coating; industrial	283.28
Point	Waste disposal	500.29
Point	Indus process; chemical manufacturing	29.30
Total		9,911

ICE = internal combustion engine; NEC = not elsewhere classified

Source: EPA 2011a

Table 6-3. 2008 National Air Emissions for Trichloroethylene

Sector	Air emissions in tons
Agriculture; livestock waste	0.02
Bulk gasoline terminals	1.93
Commercial cooking	0.00
Dust; construction dust	0.00
Fuel comb; commercial/institutional; biomass	0.35
Fuel comb; commercial/institutional; natural gas	0.09
Fuel comb; commercial/institutional; oil	0.05
Fuel comb; commercial/institutional; other	0.15
Fuel comb; electric generation; biomass	1.06
Fuel comb; electric generation; coal	0.14
Fuel comb; electric generation; natural gas	0.01
Fuel comb; electric generation; oil	0.02
Fuel comb; electric generation; other	0.07
Fuel comb; industrial boilers, ICEs; biomass	5.07
Fuel comb; industrial boilers, ICEs; coal	0.04
Fuel comb; industrial boilers, ICEs; natural gas	56.47
Fuel comb; industrial boilers, ICEs; oil	0.43
Fuel comb; industrial boilers, ICEs; other	0.17
Gas stations	0.03
Industrial processes; cement manuf	0.06
Industrial processes; chemical manuf	20.58
Industrial processes; ferrous metals	2.33
Industrial processes; mining	0.01
Industrial processes; NEC	248.05
Industrial processes; non-ferrous metals	17.88
Industrial processes; petroleum refineries	8.75
Industrial processes; pulp and paper	28.65
Industrial processes; storage and transfer	15.22
Miscellaneous non-industrial NEC	0.00
Mobile; non-road equipment; diesel	0.33
Solvent; consumer and commercial solvent use	164.60
Solvent; degreasing	2,781.51
Solvent; dry cleaning	0.07
Solvent; graphic arts	7.06
Solvent; industrial surface coating and solvent use	197.43
Waste disposal	133.18
Total	3,692

ICE = internal combustion engine; NEC = not elsewhere classified

Source: EPA 2013b

Table 6-4. 2011 National Air Emissions for Trichloroethylene

Sector	Air emissions in tons
Agriculture; livestock waste	0.002
Bulk gasoline terminals	0.04
Fuel comb; commercial/institutional; biomass	0.15
Fuel comb; commercial/institutional; natural gas	0.00009
Fuel comb; commercial/institutional; oil	0.0005
Fuel comb; commercial/institutional; other	0.16
Fuel comb; electric generation; biomass	1.28
Fuel comb; electric generation; coal	0.16
Fuel comb; electric generation; natural gas	0.09
Fuel comb; electric generation; oil	0.02
Fuel comb; electric generation; other	0.23
Fuel comb; industrial boilers, ICEs; biomass	5.6
Fuel comb; industrial boilers, ICEs; coal	0.07
Fuel comb; industrial boilers, ICEs; natural gas	1.74
Fuel comb; industrial boilers, ICEs; oil	0.043
Fuel comb; industrial boilers, ICEs; other	0.08
Industrial processes; cement manuf	0.24
Industrial processes; chemical manuf	63.6
Industrial processes; ferrous metals	15.9
Industrial processes; NEC	386.3
Industrial processes; non-ferrous metals	55.08
Industrial processes; petroleum refineries	4.33
Industrial processes; pulp and paper	33.06
Industrial processes; storage and transfer	8.18
Solvent; consumer and commercial solvent use	365.8
Solvent; degreasing	2162.70
Solvent; dry cleaning	0.008
Solvent; graphic arts	8.35
Solvent; industrial surface coating and solvent use	136.29
Total	3,250

ICE = internal combustion engine; NEC = not elsewhere classified

Source: EPA 2015b

trichloroethylene releases in the global atmosphere suggested much lower levels than are actually observed in the southern hemisphere, far from areas of release, indicating that oceanic emissions may be important (Olaguer 2002).

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). Pratt et al. (2004) estimated that 87 metric tons of trichloroethylene were released from publicly owned treatment works located in the Minneapolis-St. Paul, Minnesota metropolitan area in 1999.

6.2.2 Water

Estimated releases of 34 pounds (0.02 metric tons) of trichloroethylene to surface water, waste water treatment sites (metals only), and publicly owned treatment works (POTWs) (metal and metal compounds) from 154 domestic manufacturing and processing facilities in 2017, accounted for <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI17 2018). These releases are summarized in Table 6-1.

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).

6.2.3 Soil

Estimated releases of 115,793 pounds (52.5 metric tons) of trichloroethylene to soils from 154 domestic manufacturing and processing facilities in 2017, accounted for about 5.7% of the estimated total environmental releases from facilities required to report to the TRI (TRI17 2018). An additional 3,723 pounds (1.7 metric tons), were released via Class I wells, Class II-V wells, and underground injection (TRI17 2018). These releases are summarized in Table 6-1.

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).

6.3 ENVIRONMENTAL FATE

6.3.1 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 6.4.2). 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 9.85×10^{-3} atm-m^3/mol at 25°C suggests that trichloroethylene partitions rapidly to the atmosphere from surface water. The major route of removal of trichloroethylene from water is volatilization (EPA 1985c). 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).

Based on the measured Henry's law constant, the estimated volatilization half-life of trichloroethylene is 1.2 hours from a model river (1 meter deep, flowing 1 meter per second, with a wind velocity of 5 meters per second) and 4.6 days from a model lake (1 meter deep, flowing 0.05 meters per second, with a wind velocity of 0.5 meters per second) (EPA 2010b). 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 is 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 generally range from 49 to 460 (Brigmon et al. 1998; Chiou and Kile 1998; Garbarini and Lion 1986; Mouvet 1993; Rathbun 1998; Sahoo and Smith 1997). The components of soil organic matter show widely varying affinities for trichloroethylene, with the fats-waxes-resins fraction being responsible for stronger adsorption of trichloroethylene (Garbarini and Lion 1986; Grathwohl 1990). 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 that 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 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 (Petersen et al. 1994).

A number of groundwater monitoring studies have detected trichloroethylene in groundwater (see Section 6.4.2), 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 to moderate 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 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 <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 6.4.4), 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. Uptake of trichloroethylene in apple and peach trees and wheat, corn, and tomato seedlings has also been demonstrated (Chard et al. 2006; Doucette et al. 2007; Su et al. 2010).

6.3.2 Transformation and Degradation

6.3.2.1 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 (2.36×10^{12} cm^3/molecule-second) and a typical atmospheric hydroxyl radical concentration (5×10^5 molecules/cm^3) (Atkinson 1985, 1989), the half-life can be estimated to be 6.8 days. Class and Ballschmiter (1986) state it as between 3 and 7 days.

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 1985c). 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).

6.3.2.2 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). In addition, the photolysis rate was found to increase when the initial concentration of trichloroethylene was low (Dobaradarane et al. 2012). The study also concluded that pH did not affect the photolysis rate of trichloroethylene.

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 1.3×10^6 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.

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 4th 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 (Dong et al. 2009; Parsons et al. 1985; Wilson et al. 1986). Squillace and Moran (2007) reported that concentrations of trichloroethylene in oxic groundwater sampled across the United States were approximately 2 orders of magnitude larger than in anoxic groundwater and cited a slower rate of biodegradation under oxic conditions. 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).

6.3.2.3 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 ^{13}C nuclear magnetic resonance (NMR) spectra showed no evidence of chemical reactions (Jurkiewicz and Maciel 1995). Because trichloroethylene is a dense nonaqueous 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 a. 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 yield 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 that varied with soil type, temperature, and initial concentration of trichloroethylene (Yagi et al. 1992). The lotus field soils degraded >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}C-trichloroethylene to radiolabelled carbon dioxide ($^{14}CO_2$) 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 hydrogen as the electron donor has been isolated (Maymo-Gatell 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; 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 that 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.

6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to trichloroethylene depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of trichloroethylene in unpolluted atmospheres and in pristine surface waters are often so low as to be near the limits of current analytical methods. In reviewing data on trichloroethylene levels monitored or estimated in the environment, please note that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable. The analytical methods available for monitoring trichloroethylene in a variety of environmental media are detailed in Chapter 7.

6.4.1 Air

Trichloroethylene is widely detected in ambient air. The U.S. EPA maintains the Air Quality System (AQS) database, which serves as a repository of criteria air pollutant and HAPs monitoring data from the United States, Puerto Rico, and the Virgin Islands. Tables 6-5 and 6-6 illustrate the annual mean percentile distributions of trichloroethylene from hundreds of monitoring sites for years 1998–2008 and 2010–2015, respectively. In general, most measured values for these years are <1 ppb; however, a measured value of 164 ppb was recorded in Lake County, Indiana on November 6, 2013 and a value of 49 ppb was recorded on May 3, 2011 in Klamath Falls, Oregon (EPA 2015a), which suggests that trichloroethylene levels are highly variable in the atmosphere depending upon whether or not there are significant point sources nearby.

The data presented in Tables 6-5 and 6-6 suggest a general decline in ambient air levels of trichloroethylene over the past 2 decades and are consistent with an analysis performed by McCarthy et al. (2007) of AQS data over three trend periods: 1990–2005, 1995–2005, and 2000–2005, which suggested a decrease of about 4–7% for the median trichloroethylene levels annually.

Table 6-5. Percentile Distribution of Annual Mean Trichloroethylene Concentrations (ppb) Measured in Ambient Air at Locations Across the United States[a]

Year	Number of U.S. locations	25th	50th	75th	95th	Maximum
1998	132	0.025	0.030	0.046	0.141	1.027
1999	170	0.025	0.030	0.050	0.158	0.815
2000	187	0.025	0.030	0.050	0.196	1.375
2001	205	0.021	0.025	0.039	0.097	2.399
2002	259	0.016	0.025	0.045	0.250	3.427
2003	250	0.025	0.030	0.050	0.250	1.287
2004	264	0.025	0.025	0.050	0.210	1.075
2005	328	0.025	0.027	0.048	0.179	1.234
2006	298	0.015	0.025	0.035	0.127	1.064
2007	317	0.016	0.025	0.030	0.091	0.750
2008	288	0.012	0.025	0.030	0.108	1.145

[a] Data originally reported in units parts per billion carbon but converted to ppb volume in the table.

Source: EPA 2011f

Table 6-6. Percentile Distribution of Annual Mean Trichloroethylene Concentrations (ppb) Measured in Ambient Air at Locations Across the United States[a]

Year	Number of U.S. locations	25th	50th	75th	95th	Maximum
2010	540	0.0028	0.021	0.13	0.16	1.24
2011	518	0.0017	0.019	0.12	0.15	3.14
2012	512	0.0018	0.015	0.12	0.15	1.08
2013	477	0.0015	0.015	0.12	0.15	3.19
2014	444	0.0011	0.015	0.12	0.14	0.52
2015	220	0.0000	0.0010	0.0033	0.018	0.24
2016	209	0.0000	0.0009	0.0055	0.023	0.1035
2017	178	0.0000	0.0000	0.0003	0.0099	0.0607
2018	159	0.0000	0.0000	0.0017	0.0128	0.0714

[a] Data originally reported in units parts per billion carbon but converted to ppb volume in the table.

Source: EPA 2018

Historical monitoring data for trichloroethylene in ambient air in the United States, prior to 1981, were compiled by EPA (1982d). 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 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 μg/m^3 (0.04–0.72 ppb) in Portland, Oregon, in 1984 (Ligocki et al. 1985); 2.1 μg/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 0.096–0.225 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). The mean and median concentrations of trichloroethylene in air samples collected at 25 sites across Minnesota between 1991 and 1998 were 0.43 and 0.21 μg/m^3 (0.08 and 0.04 ppb), respectively (Pratt et al. 2000); the reported lower detection limit was 0.04 μg/m^3. In the studies that discuss seasonal variation, 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 6.3.2.1).

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).

Some elevated outdoor air levels of trichloroethylene 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).

A survey of indoor air showed median concentrations of trichloroethylene as high as 27 μg/m^3 (5.0 ppb) in a North Carolina office building; 0.74 μg/m^3 (0.14 ppb) in a Washington, DC, school; and 0.82 μg/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). Gordon et al. (1999) detected trichloroethylene in the indoor air of two out of 185 homes in Arizona with a maximum concentration of 24 µg/m^3 (4.39 ppb); the detection limit was 1.8 µg/m^3 (0.33 ppb). Weisel et al. (2008) detected trichloroethylene in the air of 8 out of 100 homes located in both suburban and rural areas of New Jersey with maximum and 95th percentile values of 13 and 2.74 µg/m^3 (2.38 and 0.50 ppb), respectively; the detection limit was reported to be approximately 1 µg/m^3 (0.18 ppb). McKone (1987) estimated the levels of trichloroethylene in indoor air based on the properties of trichloroethylene and a three-compartment model. If the tap water contained 1 mg trichloroethylene/L (1,000 ppm), then the air in the shower during use was estimated to contain 0.018 mg/L (3.3 ppm), while the airborne trichloroethylene concentration in the rest of the house was estimated to be 0.11 mg/m^3 (0.02 ppm) during the day (7 am–11 pm) and 0.024 mg/m^3 (0.0044 ppm) during the night (11 pm–7 am).

Loh et al. (2006) measured geometric mean and maximum trichloroethylene concentrations of 0.43 and 115 µg/m^3 (0.08 and 21.1 ppb), respectively, in the air of over 100 stores in the greater Boston, Massachusetts area and 0.23 and 118 µg/m^3 (0.04 and 21.6 ppb), respectively, in the air of 20 dining establishments in the same region. Martin et al. (2005) measured the concentrations of trichloroethylene at buildings located near an industrial facility in Clark County, Georgia. The method limit of detection was 25 ng/m^3 (0.0046 ppb). Levels in indoor air were 0.92 µg/m^3 (0.17 ppb) at an elementary school, 0.59–1.85 µg/m^3 (0.11–0.34 ppb) at a local business, and 0.21–4.66 µg/m^3 (0.04–0.85 ppb) in three homes. Levels measured in outdoor air at these same locations were 0.72 µg/m^3 (0.13 ppb), 1.30–4.59 µg/m^3 (0.24–0.84 ppb), and 0.03–0.05 µg/m^3 (0.005–0.009 ppb), respectively.

Mean concentrations of trichloroethylene measured during the Minnesota Children’s Pesticide Exposure Study (MCPES) were 0.8 µg/m^3 (0.15 ppb) in 73 personal air samples, 0.6 µg/m^3 (0.11 ppb) in 101 indoor air samples, and 0.6 µg/m^3 (0.11 ppb) in 100 outdoor air samples collected from households with children; the mean detection limit was 0.2 µg/m^3 (0.037 ppb) (Adgate et al. 2004a). During the School Health Initiative: Environment, Learning, Disease (SHIELD) study, the concentrations of trichloroethylene were measured in the outdoor home air, indoor school air, indoor home air, and personal air of 113 children from two inner-city schools in Minneapolis, Minnesota (Adgate et al. 2004b). Median concentrations of trichloroethylene in each of these sampling groups during both summer and winter were between 0.1 and 0.3 µg/m^3 (0.02–0.05 ppb); between 56 and 90% of the samples were above an unspecified detection limit. The mean concentrations of trichloroethylene measured in the winter during the Toxic Exposure Assessment, Columbia/Harvard (TEACH) study were 0.36 µg/m^3 (0.07 ppb) in home outdoor air, 1.26 µg/m^3 (0.23 ppb) in home indoor air, and 2.62 µg/m^3 (0.48 ppb) in personal air associated with 36 students from west central Harlem in New York City (Kinney et al. 2002). Trichloroethylene concentrations measured in the summer were 0.24 μg/m^3 (0.04 ppb) in home outdoor air, 0.32 μg/m^3 (0.06 ppb) in home indoor air, and 0.51 μg/m^3 (0.09 ppb) in personal air associated with 31–40 students from the same area. The limit of detection was 0.15 μg/m^3 (0.027 ppb). Clayton et al. (1999) reported mean trichloroethylene levels of 5.27 μg/m^3 (0.96 ppb) in 386 personal air samples, 39.4% of which were above an unspecified limit of detection, 2.84 μg/m^3 (0.52 ppb) in 402 indoor air samples (36.1% above the limit of detection), and 1.11 μg/m^3 (0.20 ppb) in 97 outdoor air samples (26.4% above the limit of detection) collected in EPA Region 5.

Brenner (2010) measured median and maximum trichloroethylene concentrations of 0.895 and 1.69 μg/m^3 (0.16 and 0.31 ppb), respectively, for 541 indoor air samples from four large buildings at the NASA Ames Research Center at the southern end of San Francisco Bay; the frequency of detection was 99.8%. The levels were attributed to vapor intrusion from underlying contaminated groundwater. Vapor intrusion is the process whereby trichloroethylene may evaporate from contaminated groundwater and soil and migrate into air spaces beneath buildings to enter the indoor air (EPA 2002). Burk and Zarus (2013) reported selected results from 135 vapor intrusion public health assessments and consultations for 121 sites published on ATSDR’s website between 1994 and 2009. Trichloroethylene indoor air levels were attributed to vapor intrusion and detected at 21 sites; levels at 3 of these sites were high enough to be considered a public health hazard.

Sapkota et al. (2005) reported median and maximum trichloroethylene concentrations of 3.11 and 6.89 μg/m^3 (0.57 and 1.26 ppb), respectively, in the indoor air of a tollbooth at the Baltimore Harbor Toll Plaza and 0.06 and 0.56 μg/m^3 (0.01 and 0.10 ppb), respectively, in the air outside the tollbooth.

6.4.2 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 part per trillion (ppt) (Sauer 1981). Average levels of 7 ng/L (7 ppt), 0.3 ppt, and 74 ng/L (74 ppt) were found in the northeastern Atlantic (Murray and Riley 1973), Liverpool Bay (Pearson and McConnell 1975), and the southern North Sea (Huybrechts et al. 2005), respectively.

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 (2011d) released the results of its second 6-year review of 69 regulated contaminants in public water systems (PWS) located across the United States. Data for trichloroethylene are summarized in Table 6-7. During 2005, trichloroethylene was detected in 2,292 out of 46,937 samples (4.9%) collected from groundwater supplied PWS and 1,874 out of 12,705 samples (14.8%) collected from surface water supplied PWS. The median, 95th percentile, and maximum concentrations of the positive samples were 1.1, 13.0, and 159 ppb, respectively, in groundwater supplied PWS and 1.6, 28.0, and 50.0 ppb, respectively, in the surface water supplied PWS. Rowe et al. (2007) detected trichloroethylene in 41 out of 1,207 U.S. domestic well samples collected between 1996 and 2002, a detection frequency of 3.4%. 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 (ppb), with a single maximum level of 130 μg/L (ppb) (Westrick et al. 1984).

Williams et al. (2002) reported annual levels of trichloroethylene measured in 3,447–4,226 California drinking water sources between 1995 and 2001. Trichloroethylene was detected in 9.6–11.7% of the sources over the time period with an average detected concentration ranging from 14.2 to 20.7 µg/L (ppb). Trichloroethylene was detected in groundwater samples from approximately 55% of 30 public supply wells and 10% of 95 monitoring wells located in a region of southern New Jersey (Stackelberg et al. 2001). The maximum concentrations of trichloroethylene measured in community water systems near Dayton, Ohio during 2004 were 3.29 µg/L (ppb) in source water and 0.21 µg/L (ppb) in finished water (Rowe et al. 2007). 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).

Drinking water supplies at Camp Lejeune have been shown to be heavily contaminated with trichloroethylene and other chlorinated solvents due to handling and disposal practices of an off-site dry cleaning facility (ATSDR 2017b). Water samples obtained from the Hadnot Point Water Treatment plant at Camp Lejeune had levels of trichloroethylene of up to 1,400 µg/L in 1982 (ATSDR 2017b). A recent historical reconstruction study of this site, which applied additional modeling methods, reported trichloroethylene concentrations at the Hadnot Point water treatment plant reached a maximum monthly average value of 783 µg/L compared to a one-time maximum measured value of 1,400 µg/L during the period of August 1953 to December 1984 (Maslia et al. 2016).

Table 6-7. Concentrations of Trichloroethylene (ppb) Measured in Drinking Water from Public Water Systems (PWS) Across the United States

	Year	Total samples	Positive detections[a]	Mean[b]	Median[b]	95th Percentile[b]	Maximum
Groundwater supplied PWS							
	1998	29,356	1,308 (4.5%)	2.99	1.20	11.0	39.3
	1999	29,695	1,383 (4.6%)	3.05	1.30	11.8	47.0
	2000	32,794	1,545 (4.7%)	2.76	1.20	9.00	49.0
	2001	36,657	1,713 (4.7%)	3.55	1.30	13.0	71.6
	2002	43,934	2,119 (4.8%)	3.72	1.40	13.0	58.7
	2003	42,466	2,199 (5.2%)	3.67	1.60	13.8	58.0
	2004	46,490	2,283 (4.9%)	3.70	1.30	15.0	90.9
	2005	46,937	2,292 (4.9%)	3.33	1.10	13.0	159
Surface water supplied PWS							
	1998	9,020	1,628 (18%)	5.69	2.50	29.0	50.0
	1999	11,415	1,877 (16%)	6.09	2.50	25.3	50.0
	2000	11,521	1,696 (15%)	5.70	2.00	28.3	50.0
	2001	11,913	2,085 (18%)	4.71	1.46	19.9	50.0
	2002	13,287	1,926 (14%)	4.86	1.60	20.1	50.0
	2003	12,605	1,813 (14%)	4.43	1.50	19.4	50.0
	2004	12,814	1,816 (14%)	4.80	1.80	22.0	50.0
	2005	12,705	1,874 (15%)	5.40	1.60	28.0	50.0

[a] Limit of detection for most samples was 0.5 ppb.
[b] Mean, median, and 95th percentile concentrations of positive detections.

Source: EPA 2011d

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) (Burmaster 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 5 of 6 Minnesota municipal solid waste landfills at levels ranging from 0.7 to 125 μg/L (ppb) and in groundwater near 9 of 13 landfills at levels ranging from 0.2 to 144 μg/L (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 (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, New Hampshire, 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 0.0059–160 mg/L (ppm); detection limits were <0.0005 mg/L (ppm). Average trichloroethylene concentrations measured in groundwater at different locations at a Superfund site (former auto parts manufacturing) located on the shore of Lake Michigan ranged from 14.6 to 6,554 μg/L (ppb) (An et al. 2004). The average concentration measured in sediment water collected 100 m offshore from the site was 1.37 μg/L (ppb); the reported detection limit for chlorinated ethenes was 0.5 μg/L (ppb). Brusseau et al. (2007) reported concentrations of trichloroethylene ranging from 100 to approximately 12,000 μg/L (ppb) in groundwater collected at the Tucson International Airport Area federal Superfund site in Southern Arizona.

An analysis of the EPA STORET 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). Data from January through December of 2015 indicated that trichloroethylene was detected in 5 out of 298 river/stream samples in the United States at levels ranging from 0.20 to 8.6 μg/L (ppb) (EPA 2015c). 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 trichloroethylene levels of 0.008 and 0.13 μg/L (ppb) were reported for water samples collected from the lower Niagara River and Lake Ontario in 1981 (Strachan and Edwards 1984); trichloroethylene was detected in 12/17 samples from the Niagara River and 16/83 samples from Lake Ontario. Asher et al. (2007) measured trichloroethylene concentrations ranging from 0.15 to 0.32 μg/L (ppb) in a section of the Aberjona River near Woburn, Massachusetts; the limit of detection was 0.02 μg/L (ppb).

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).

Monitoring wells at 30 sites in Salt Lake Valley, Utah were used to analyze VOCs including trichloroethylene). Trichloroethylene was detected in five of the wells at maximum concentration of 1.54 μg/L (ppb) and a minimum concentration of 0.02 μg/L (ppb). These values were both under the maximum contaminant level of 5 μg/L (ppb) (USGS 2003).

Samples from 2,401 domestic and 1,096 public wells across the nation were monitored for VOCs. Approximately 3,500 water samples were collected during 1985–2001. Concentrations of human-health concern (defined in the report as concentrations greater than an EPA MCL or concentrations greater than the health-based screening level) for trichloroethylene were found in both domestic and public wells. Trichloroethylene was the 4th most frequently detected VOC in the study and had the highest median concentration of all VOC detections. At an assessment level of 0.2 μg/L (ppb), the overall detection frequency for trichloroethylene was about 3% and was higher in the public well samples than in the domestic samples, approximately 4 and 1%, respectively. At an assessment level of 0.02 μg/L (ppb), the detection frequency was close to 4%. It was noted that concentrations with potential concern could not be directly related to concentrations in drinking water that would cause concern due to the nature of the sampling. (USGS 2006).

6.4.3 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, New Hampshire (Hewitt and Shoop 1994). Concentrations of trichloroethylene in soil from the saturated zone were 0.008–25 mg/kg (ppb), while concentrations in the groundwater were 44–180,000 ppb.

Soil gas, at three former fuel-dispensing sites in Fort Gorgon, Georgia, was monitored from October 2010 to September 2011 (USGS 2012). Fifty-five soil-gas samplers were installed at one site, 30 samplers at another site, and 39 samplers at the third site. The method detection limit was 0.02 μg/L (ppb). Trichloroethylene was not detected in the majority of the samples; however, it was detected just below the detection limit in two of the samples and detected at a mass of 0.05 μg in one of the samples (USGS 2012).

6.4.4 Other Environmental Media

Trichloroethylene was detected in 30 table-ready food items collected from supermarkets across the United States during a 5-year study (1996–2000) conducted by the U.S. Food and Drug Administration (Fleming-Jones and Smith 2003). Minimum and maximum concentrations are listed in Table 6-8. Reported concentrations were between 2 and 10 ppb in most items. However, maximum levels were much higher in beef frankfurters (105 ppb), chocolate cake with icing (57 ppb), raw avocado (75 ppb),

Table 6-8. Levels of Trichloroethylene Measured in Table-Ready Foods from Across the United States During a 5-Year Study (1996–2000)

Type of food	Positive detections	Minimum (ppb)	Maximum (ppb)
American cheese	2	2	2
Cheddar cheese	1	2	2
Mixed nuts	2	2	5
Ground beef	2	3	6
Banana raw	1	2	2
Cream cheese	2	2	3
Frankfurters, beef	5	2	105
Chocolate cake with icing	3	3	57
Tuna, canned in oil	2	9	11
Fruit-flavored cereal	1	3	3
Peanut butter	3	4	70
Avocado, raw	6	2	75
Popcorn, popped in oil	2	4	8
Blueberry muffin	2	3	4
Orange, raw	1	2	2
Coleslaw with dressing	1	3	3
Sweet roll/danish	3	3	4
Potato chips	4	4	140
Quarter pound hamburger, cooked	2	5	9
Margarine	3	2	21
Butter	2	7	9
Chocolate chip cookies	2	2	4
Apple pie, fresh/frozen	2	2	4
Chicken nuggets, fast food	3	2	5
French fries, fast food	2	3	3
Cheeseburger, quarter pound	1	7	7
Cheese pizza	1	2	2
Bologna	5	2	20
Cheese and pepperoni pizza	2	2	2
Cake doughnuts with icing	1	3	3

Source: Fleming-Jones and Smith 2003

and potato chips (140 ppb). McConnell et al. (1975) reported the detection of trichloroethylene in dairy products (milk, cheese, butter) at 0.3–10 μg/kg (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. 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 ng/g (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 that exceed acceptable limits (Grob et al. 1990). The authors estimated that in order to limit food concentrations of trichloroethylene to 50 μg/kg (ppb; the maximum tolerated limit for food halocarbons in Switzerland), the level in surrounding air should not exceed 38.5 μg/m^3 (7 ppb). 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 8).

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

6.5 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 and 0.763 ng/L (763 ppb), 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 (ppb). The concentrations in drinking water in the city were <0.05–22.93 and 0.21–7.80 μg/L (ppb) for trichloroethylene and tetrachloroethylene, respectively.

The Fourth National Report on Human Exposure to Environmental Chemicals (CDC 2018) includes results from assessment of trichloroethylene levels in blood samples from 923 cigarette smokers and 2,054 nonsmokers within the U.S. general population surveyed during the years 2013 and 2014 (Table 6-9). Trichloroethylene was below the detection limit of 0.012 ng/mL (ppb).

Assuming a typical air concentration range of 100–500 ppt (Singh et al. 1981, 1982) and a breathing rate of 20 m^3 air/day, the average daily air intake of trichloroethylene can be estimated at 11–33 μg/day. Average daily water intake of trichloroethylene can be estimated at 2–20 μg/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^3 (93–15,072 ppb) in <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 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

Table 6-9. Geometric Mean and Selected Percentiles of Blood Concentrations (in ng/mL) for the U.S. Population from the National Health and Nutrition Examination Survey

	Survey years[a]	Geometric mean (95% CI)	Selected percentiles (95% CI)				Sample size
			50th	75th	90th	95th	
Total	01–02	*[b]	<LOD[c]	<LOD	<LOD	<LOD	922
	03–04	*	<LOD	<LOD	<LOD	<LOD	1,228
	05–06	*	<LOD	<LOD	<LOD	<LOD	3,178
	07–08	*	<LOD	<LOD	<LOD	<LOD	2,952
	09–10	*	<LOD	<LOD	<LOD	<LOD	3,342
	11–12	*	<LOD	<LOD	<LOD	<LOD	2,694
	13–14	*	<LOD	<LOD	<LOD	<LOD	3,103
Age group							
12–19 years	05–06	*	<LOD	<LOD	<LOD	<LOD	937
	07–08	*	<LOD	<LOD	<LOD	<LOD	473
	09–10	*	<LOD	<LOD	<LOD	<LOD	568
	11–12	*	<LOD	<LOD	<LOD	<LOD	496
	13–14	*	<LOD	<LOD	<LOD	<LOD	581
20–59 years	01–02	*	<LOD	<LOD	<LOD	<LOD	922
	03–04	*	<LOD	<LOD	<LOD	<LOD	1,228
	05–06	*	<LOD	<LOD	<LOD	<LOD	1,562
	07–08	*	<LOD	<LOD	<LOD	<LOD	1,592
	09–10	*	<LOD	<LOD	<LOD	<LOD	1,840
≥20 years	11–12	*	<LOD	<LOD	<LOD	<LOD	2,198
	13–14	*	<LOD	<LOD	<LOD	<LOD	2,522
≥60 years	05–06	*	<LOD	<LOD	<LOD	<LOD	679
	07–08	*	<LOD	<LOD	<LOD	<LOD	887
	09–10	*	<LOD	<LOD	<LOD	<LOD	934
Gender							
Males	01–02	*	<LOD	<LOD	<LOD	<LOD	434
	03–04	*	<LOD	<LOD	<LOD	<LOD	604
	05–06	*	<LOD	<LOD	<LOD	<LOD	1,504
	07–08	*	<LOD	<LOD	<LOD	<LOD	1,466
	09–10	*	<LOD	<LOD	<LOD	<LOD	1,647
	11–12	*	<LOD	<LOD	<LOD	<LOD	1,343
	13–14	*	<LOD	<LOD	<LOD	<LOD	1,492
Females	01–02	*	<LOD	<LOD	<LOD	<LOD	488
	03–04	*	<LOD	<LOD	<LOD	<LOD	624
	05–06	*	<LOD	<LOD	<LOD	<LOD	1,674
	07–08	*	<LOD	<LOD	<LOD	<LOD	1,486
	09–10	*	<LOD	<LOD	<LOD	<LOD	1,695
	11–12	*	<LOD	<LOD	<LOD	<LOD	1,351
	13–14	*	<LOD	<LOD	<LOD	<LOD	1,611
Race/ethnicity							
Mexican Americans	01–02	*	<LOD	<LOD	<LOD	<LOD	228
	03–04	*	<LOD	<LOD	<LOD	<LOD	224
	05–06	*	<LOD	<LOD	<LOD	<LOD	773
	07–08	*	<LOD	<LOD	<LOD	<LOD	572
	09–10	*	<LOD	<LOD	<LOD	<LOD	666
	11–12	*	<LOD	<LOD	<LOD	<LOD	284
	13–14	*	<LOD	<LOD	<LOD	<LOD	495

Table 6-9. Geometric Mean and Selected Percentiles of Blood Concentrations (in ng/mL) for the U.S. Population from the National Health and Nutrition Examination Survey

	Survey years[a]	Geometric mean (95% CI)	Selected percentiles (95% CI)				Sample size
			50th	75th	90th	95th	
Non-Hispanic blacks	01–02	*	<LOD	<LOD	<LOD	<LOD	191
	03–04	*	<LOD	<LOD	<LOD	<LOD	266
	05–06	*	<LOD	<LOD	<LOD	<LOD	828
	07–08	*	<LOD	<LOD	<LOD	<LOD	583
	09–10	*	<LOD	<LOD	<LOD	<LOD	598
	11–12	*	<LOD	<LOD	<LOD	<LOD	724
	13–14	*	<LOD	<LOD	<LOD	<LOD	600
Non-Hispanic whites	01–02	*	<LOD	<LOD	<LOD	<LOD	441
	03–04	*	<LOD	<LOD	<LOD	<LOD	644
	05–06	*	<LOD	<LOD	<LOD	<LOD	1,341
	07–08	*	<LOD	<LOD	<LOD	<LOD	1,329
	09–10	*	<LOD	<LOD	<LOD	<LOD	1,520
	11–12	*	<LOD	<LOD	<LOD	<LOD	927
	13–14	*	<LOD	<LOD	<LOD	<LOD	1,256
All Hispanics	11–12	*	<LOD	<LOD	<LOD	<LOD	572
	13–14	*	<LOD	<LOD	<LOD	<LOD	795
Asians	11–12	*	<LOD	<LOD	<LOD	<LOD	389
	13–14	*	<LOD	<LOD	<LOD	<LOD	344

[a] Survey period 2001–2002 is a one-third subsample of 20–59-year-olds; survey period 2003–2004 is a one-half subsample of 20–59-year-olds.
[b] Not calculated; the proportion of results below the LOD was too high to provide a valid result.
[c] <LOD means less than the limit of detection of 0.012 ng/mL (ppb).

CI = confidence interval; LOD = limit of detection

Source: CDC 2018

outdoor air, even near major point sources such as chemical plants (Adgate et al. 2004a, 2004b; Clayton et al. 1999; Kinney et al. 2002; 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 μg/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). Postmortem analyses of human tissue revealed body fat levels of 1.4–32 μg/kg (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 (EPA 1987j). 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).

Exposure to trichloroethylene via drinking water and the effects of pregnancy outcomes has been evaluated in residents of base family housing at Camp Lejeune in Onslow County, North Carolina (ATSDR 1998). In 1982, two drinking water supplies that were monitored over a 34-month period at Camp Lejeune were found to be contaminated with VOCs including trichloroethylene. Close monitoring was conducted as a means for the restoration program, and contamination has not been detected in these systems since February of 1985. The highest level of trichloroethylene measured in tap water samples was 1,400 ppb. The primary source of the contamination was attributed to a dry cleaning establishment.

There were 141 short-term exposed women, 31 long-term exposed women, and 5,681 unexposed women studied with regard pregnancy and mean birth weight (MBW), small for gestational age (SGA) and preterm birth. Results found that preterm delivery was not associated with exposure. A difference between the MBW of the long-term exposed women and the unexposed women was -139 g (90% confidence limit). The OR was 1.5 for SGA (90% confidence limit) and long-term exposure. For male births, the OR for SGA was 3.9 (90% confidence limit) and the difference in MBW for this group was -312 (90% confidence limit). Short-term exposed women had a lower occurrence of SGA infants and the MBW was slightly higher than infants of unexposed women. Further investigation with a larger group of participants was suggested to examine the potentially important link between reduced birth weight and SGA in male infants born to mothers who have been exposed, long-term, to trichloroethylene.

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 (NIOSH 1990). The NOES database does not contain 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 were obtained from degreasing operations, which is the primary industrial use of trichloroethylene (Bakke et al. 2007; Franco et al. 2007; Green et al. 2004; Murata et al. 2010; Raaschou-Nielson et al. 2001; Rosa 2003; von Grote et al. 2003). Worker exposure data indicated that exposure is likely to vary, although TWA concentrations were generally consistent and usually ranged from ≤50 to 100 ppm (NCI 1985). More recent occupational monitoring data from the United States were not located. von Grote et al. (2003) reviewed historical occupational exposure measurements at facilities in Europe and reported average workplace air concentrations measured between 1980 and 2000 generally between <1.0 and 15.4 ppm with 95th percentile values of <170 ppm. OSHA allows an 8-hour TWA permissible exposure limit of 100 ppm (Rosa 2003). The 15-minute TWA exposure, which should not be exceeded at any time during a workday, is 300 ppm (OSHA 1993; Rosa 2003). 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 (NCI 1985). Workers at a clock manufacturing facility located in Bangkok, Thailand where trichloroethylene was used as a degreasing agent had much higher urinary trichloroacetic acid (TCA) levels when compared to a control group (Singthong et al. 2015). Analysis of the air in the breathing zone of the manufacturing facility showed an average trichloroethylene level of 27.83±6.02 ppm. Average TCA levels in exposed workers and a nonoccupationally exposed control group of subjects was 14.84±1.62 and 2.95±0.28 mg/L, respectively.

Worker exposure in the dry cleaning industry is a notable route for exposure to trichloroethylene. Exposure was evaluated using the relationship between concentrations of trichloroethylene in urine and concentrations in air collected at the breathing zone of workers in the workplace. Forty participants between the ages of 27 and 57 included 30 dry cleaning workers and 10 occupationally non-exposed individuals as a reference. The mean values for exposure of trichloroethylene in the breathing zone were found to be 1.56, 1.75, and 2.40 mg/m^3 (0.20, 0.32, and 0.44 ppm) for sites with a dry cleaning machine capacities of 8, 12, and 18 kg, respectively. The mean value for exposure of trichloroethylene in the breathing zone for the occupationally non-exposed participants was 0.98 mg/m^3 (0.18 ppm). Mean urinary concentrations before and after work shifts were measured. Levels before work were 2.38, 5.53, and 8.18 µg/L (ppb) and levels after work were 4.46, 11.31, and 4.46 µg/L (ppb) at sites with the dry cleaning machine capacity of 8, 12, and 18 kg, respectively. For occupationally non-exposed participants, levels were 0.31 µg/L (ppb) before work and 0.29 µg/L (ppb) after work (Rastkari et al. 2011).

6.6 EXPOSURES OF CHILDREN

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

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

As with adults, children are expected to be exposed to trichloroethylene through inhalation of ambient air and ingestion of drinking water. Children may also be exposed to trichloroethylene through ingestion of food containing this substance. The potential for exposure to trichloroethylene through ingestion of dust and soil is expected to be low based on the low to moderate potential for soil adsorption and the high rate of volatilization. Trichloroethylene has been detected in breast milk, indicating that this is a potential source of exposure for infants (Pellizzari et al. 1982). Beamer et al. (2012) detected trichloroethylene in 7 of 20 (35%) breast milk samples at a maximum concentration of 6 ng/mL. It was noted that the concentration in the breast milk was correlated to the concentration of trichloroethylene in water used for bathing and laundry (Beamer et al. 2012).

During the School Health Initiative: Environment, Learning, Disease (SHIELD) study, trichloroethylene was detected in only 3–7% of 416 blood samples collected from U.S. children from two neighborhoods in Minneapolis, Minnesota between 2000 and 2001 (Sexton et al. 2005). The 99th percentile concentration was reported as 0.01–0.02 ng/mL (ppb). Trichloroethylene was detected in <10% of blood samples collected from 43 children from a third Minneapolis neighborhood between 2000 and 2002 (Sexton et al. 2006).

Contaminated groundwater serves as a source for vapor intrusion of trichloroethylene into homes and is a notable exposure route (Geng et al. 2010; Johnston and MacDonald Gibson 2011). Seventeen monitoring wells at a former chemical works site that operated between 1970 and 2004 had levels as high as 1,220 mg/L (ppm) of trichloroethylene in one well, 14.8 mg/L (ppm) in another well, and <0.2 mg/L (ppm) in all of the other wells. It was reported that exposure levels of trichloroethylene for child residents from indoor air was 2.5 mg/m^3 (460 ppb) which was 2 orders of magnitude higher than outdoor air levels. However, exposure for school children was higher in the outdoor air. In addition, for point source construction workers, the concentration of trichloroethylene in outdoor air was 3.2x10^{-4} mg/m^3 (5.9x10^{-2} ppb) (Geng et al. 2010).

Information on trichloroethylene levels in tissue, neonatal blood, cord blood, and meconium fluid is not available.

6.7 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 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). Significantly elevated indoor air levels of trichloroethylene can occur via vapor intrusion, a process whereby trichloroethylene evaporates from contaminated groundwater and soil and migrates into air spaces beneath buildings to enter the indoor air (EPA 2002).

ATSDR has estimated that as many as 1 million military and civilian staff and their families might have been exposed to trichloroethylene and other chlorinated solvents in contaminated drinking water during a period dating back to the 1950s (ATSDR 2017b). Due to a lack of monitoring data at this site prior to the 1980s, the extent and duration of exposure were unknown.

A survey of 20 brands of typographical correction fluids found that several contained ≤10% 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 (EPA 1987j).

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 Total Exposure Assessment Methodology (TEAM) 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.

EPA (2014a) recently completed a chemical risk assessment for trichloroethylene use in degreasing operations, cleaning, and arts and crafts usage. EPA (2014a) identified cancer risk concerns and short-term and long-term noncancer risk for workers and occupational bystanders at small commercial degreasing facilities that use trichloroethylene-based solvents and dry cleaning facilities that use trichloroethylene-based spotting agents. EPA (2014a) also identified short-term noncancer risks for consumers and residential bystanders from the use of trichloroethylene-containing solvent degreasers and spray-applied protective coatings.

6.8 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA, as amended, directs the Administrator of ATSDR (in consultation with the Administrator of EPA and agencies and programs of the Public Health Service) to assess whether adequate information on the health effects of trichloroethylene is available. Where adequate information is not available, ATSDR, in conjunction with NTP, is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of 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 to mean that all data needs discussed in this section must be filled. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

6.8.1 Identification of Data Needs

Physical and Chemical Properties. The physical and chemical properties of trichloroethylene are well characterized (HSDB 2013; 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 2013). 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 8). 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 substance release and off-site transfer information to the EPA. The TRI, which contains this information for 2017, became available in 2018. This database is 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). Trichloroethylene is biodegraded in water (Jensen and Rosenberg 1975; Smith and Dragun 1984) and, to a limited extent, in soil (Maymo-Gatell 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; Müller et al. 1974; Sato and Nakajima 1978), oral (DeFalque 1961; D'Souza et al. 1985; Kleinfeld 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 Abdel-Rahman 1991; Sato and Nakajima 1978; Stewart and Dodd 1964; Tsuruta 1978) exposure. All of 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 (EPA 1982d; 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; Sauer 1981), soil (Hewitt and Shoop 1994; Hunter and Bromberg 1982), and food (Entz and Hollifield 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. 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. Mean TWA concentrations generally ranging from ≤50 to 100 ppm have been reported (NCI 1985). More recent monitoring of trichloroethylene levels in air from occupational settings within the United States are needed to determine whether this range is currently relevant. More information on exposure levels for populations living in the vicinity of hazardous waste sites is also needed for estimating human exposure. This information is necessary for assessing the need to conduct health studies on these populations.

Exposures of Children. Limited data are available regarding the exposures of children to trichloroethylene. Pellizzari et al. (1982) report detection of trichloroethylene in breast milk and Sexton et al. (2005, 2006) report levels of trichloroethylene measured in the blood of children from the Minneapolis, Minnesota area. Additional information regarding the levels of trichloroethylene in these and other matrices, such as tissue, neonatal blood, cord blood, and meconium fluid, would be helpful in assessing the exposure of children to this substance. Child health data needs relating to susceptibility are discussed in Section 3.12.2, Identification of Data Needs: Children's Susceptibility.

6.8.2 Ongoing Studies

As part of the National Health and Nutrition Examination Survey (NHANES), the Division of Laboratory Sciences in the National Center for Environmental Health, Centers for Disease Control, continues to analyze human blood samples for trichloroethylene and other volatile organic compounds. These data provide estimates regarding frequency of occurrence and background levels of these compounds in the general population.

ATSDR (2011b) is conducting a survey of more than 300,000 people who lived or worked at Camp Lejeune or Camp Pendleton in the 1970s and 1980s. Additional information regarding this survey is available at the following website: http://www.atsdr.cdc.gov/sites/lejeune/health_survey.html.

Other ongoing studies pertaining to trichloroethylene have been identified and are shown in Table 6-10.

Table 6-10. Ongoing Studies on Trichloroethylene

Principal investigator	Study topic	Institution	Sponsor
Smith, MT	Techniques for detection, assessment, and evaluation of human health effects of hazardous substances, including trichloroethylene	University of California Berkeley	National Institute of Environmental Health Sciences

Source: RePORTER 2018