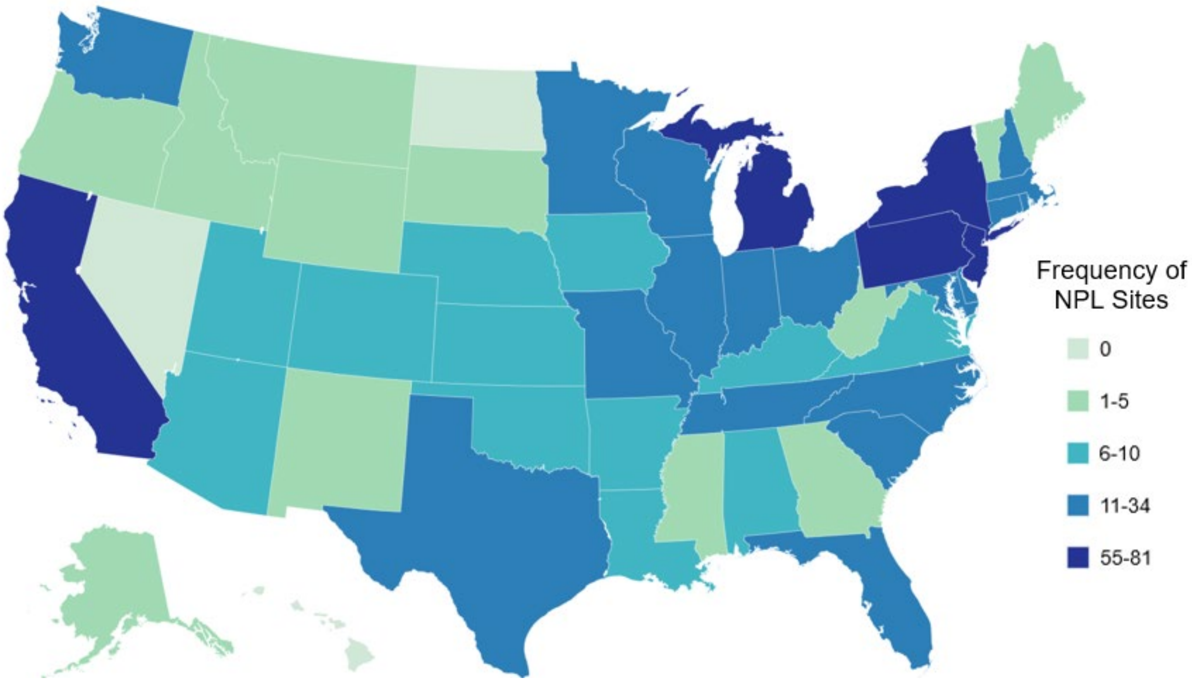
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Chloroform has been identified in at least 792 of the 1,868 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2022c). However, the number of sites in which chloroform has been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Of these sites, 783 are located within the United States, 1 is located in the Virgin Islands, and 8 are located in Puerto Rico (not shown).

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



Source: ATSDR 2022c

- The general public is most likely exposed to chloroform through ingesting food and water containing chloroform, inhaling contaminated air, and dermal contact with chloroform-containing water.
- The primary route of exposure is ingestion from the small amount of chloroform produced in drinking water as a byproduct of chlorination.
- Populations working and/or living near industries that use or create chloroform or hazardous waste sites may have an increased risk of exposure.

- Chloroform is released into the environment from industrial facility waste streams, primarily those that manufacture hydrochlorofluorocarbon-22 (HCFC-22), and as a byproduct of water disinfection.
- Chloroform is produced naturally through biotic and abiotic processes in terrestrial and aquatic environments, and as a result, is often detected in small amounts in remote environments.
- Chloroform is expected to exist almost entirely in the vapor phase in the atmosphere and is expected to volatize rapidly from surface water. Chloroform does not adsorb significantly to soil or sediment and therefore may migrate to groundwater. Chloroform does not significantly bioconcentrate in aquatic environments.
- The dominant degradation process of chloroform in the environment is the reaction of chloroform with free radicals in the atmosphere. At low concentrations and anaerobic conditions, microbial degradation of chloroform can also occur.

Chloroform is a dense liquid with a low boiling point, existing in its vapor form at temperatures above approximately 61°C. In liquid form, it is used primarily in the production of chlorodifluoromethane (HCFC-22). HCFC-22 was previously used as a refrigerant for home air conditioners or large supermarket freezers. However, as a result of the Montreal protocol and the phaseout of HCFC-22 as a refrigerant between 2010 and 2020, the demand for chloroform in the United States as a refrigerant has declined. Despite this phaseout, demand for chloroform remains high due to the use of HCFC-22 as an intermediate for fluoropolymers. Chloroform has also been used as a solvent in the pharmaceutical industry, as a heat transfer medium in fire extinguishers, as an intermediate in the preparation of dyes and pesticides, as well as in various other applications. Chloroform was previously used as a medical anesthetic, but medical use was largely phased out with availability of safer alternatives. It may still have limited medical uses in some dental procedures and in the administration of drugs for the treatment of certain diseases. Chloroform is not currently reported as an active or inert ingredient in registered pesticide products, and its use in drug, cosmetic, and food packaging products has been discontinued.

Chloroform is both a synthetic and naturally occurring compound, and natural formation may contribute a significant portion of emissions. Chloroform is released into the environment from manufacture and use. Chloroform is formed when drinking water, municipal and industrial wastewater, or swimming pool and spa water are chlorinated, or when other water treatment processes involve chlorination. Most of the chloroform released into the environment will eventually enter the atmosphere. In the atmosphere, chloroform may be transported long distances before ultimately being degraded by indirect photochemical reactions with free radicals such as hydroxyl radicals, with a half-life on the order of months. The compound has been detected in ambient air in locations that are remote from anthropogenic sources, possibly due to natural formation via abiotic or biotic processes. Chemical hydrolysis is not a significant removal process. While microbial biodegradation can take place, such reactions are generally possible only at fairly low concentration levels due to chloroform's toxicity. Microbial biodegradation of chloroform may also be inhibited due to high levels of aromatics (e.g., toluene), chlorinated hydrocarbons (e.g., trichloroethylene [TCE]), or heavy metals (e.g., zinc). Because of its low soil adsorption and water solubility, chloroform will readily leach from soil into groundwater. In groundwater, chloroform is expected to be persistent when oxygenated conditions are present.

The general population is exposed to chloroform by ingesting water and food, inhaling contaminated air, and through dermal contact with chloroform-containing water. Generalizations can be made concerning the chloroform concentrations in the environment. Background air concentrations appear to be in the parts per trillion (ppt) range, but certain urban, indoor, and source-dominated areas may show elevated concentrations when compared to background concentrations. Chlorine is the most commonly used disinfectant for drinking water treatment in the United States, reportedly used by 70% of water systems (AWWA Disinfection Committee 2021); as a result, chloroform is prevalent in tap water throughout the country. The EPA designated a maximum contaminant level goal of <70 ppb (0.07 mg/L) for chloroform (EPA 2022). Drinking water levels as high as 75 ppb have been reported in public water supplies (USGS 2006), although most of the reported concentrations are <25 ppb, typically ranging between <0.2 and 23 ppb. Levels in drinking water derived from groundwater contaminated with leachate from landfills and hazardous waste sites can sometimes be much higher. Except for a few special surveys, regular testing for chloroform or other trihalomethanes has focused on larger community water treatment systems serving at least 10,000 people. Very limited information was located regarding the concentrations found in ambient soil. Chloroform has also been detected in the ppb range in certain foods.

Occupational exposure to higher than background levels of chloroform can be expected to occur in some occupations although few quantitative exposure data were located. Populations with the highest potential exposures appear to be workers who manufacture or use chloroform and operators at incinerators, wastewater facilities, paper, or pulp plants. People who live near these facilities or contaminated hazardous waste sites may also have increased potential for exposure. Persons who derive their drinking water from groundwater sources contaminated with leachate from hazardous waste sites may also have increased exposure to chloroform, compared to the general population.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Chloroform can form naturally through biotic and abiotic mechanisms in aquatic and terrestrial environments, such as oceans, volcanoes, forest soils, grasslands, dry swamplands, peat moorland, and rice fields (Gron et al. 2012; Hoekstra et al. 2001; Laturnus et al. 2002). Chloroform flux from termite mounds has also been measured, and the concentration of chloroform within the mounds were 1,000 times greater than levels in the ambient air (Laturnus et al. 2002). Chloroform formation in soils can occur through biologically mediated chlorination of natural organic material with hypochlorous acid or other oxidized chlorine species, to an intermediate trichloroacetyl-containing compound, which is then hydrolyzed to chloroform (Breider et al. 2013). Fungi have also been shown to play a part in natural chloroform formation, which may be due to their chloroperoxidase enzymes, which catalyze the production of hypochlorous acid from chloride ions and hydrogen peroxide (Hoekstra et al. 2001; USGS 2004).

Chloroform can be formed abiotically via the decarboxylation of trichloroacetic acid by reduction of tetrachloromethane in iron-reducing environments, which may be mineral-mediated, or by the oxidation of organic matter by an electron acceptor like iron (III) (Laturnus et al. 2002). Natural formation of chloroform in water occurs through reaction of dissolved chlorine with sediment and other materials, from biological production by marine algae, and by the reaction of chlorinated pollutants with humic materials (EPA 1985a; Laturnus et al. 2002).

Chloroform is also formed in the process of making paper and as a disinfection byproduct of chlorination, a process that is used to produce potable water or treat wastewater (Ohligschläger et al. 2019). Chloroform has also been detected as a byproduct of chloride-containing cleaning products, such as hypochlorite (bleach), and can be generated when bleach is mixed with other common household chemicals, such as isopropyl alcohol (Bruchard et al. 2023; Odabasi 2008; Lin et al. 2022). Chloroform forms through the oxidation of dissolved organic material by chlorine, hypochlorite, or ozone (in the presence of chloride ions). Formation is increased when there are increased levels of free available chlorine, increased pH, increased temperature, and increased presence of organic matter (Kanan et al. 2015).

Industrially, chloroform can be produced from the chlorination of methane, methyl chloride, or methylene chloride, or from the hydrodechlorination of carbon tetrachloride (Holbrook 2003). Compounds with ketone or alcohol groups produce chloroform upon reaction with chlorine and an alkali, or hypochlorite. Methyl chloride chlorination has been reported as the most common commercial process for chloroform production (Holbrook 2003). This process is carried out in the gas phase at 400–500°C, with methyl chloride and gaseous chlorine, through free-radical reactions. Novel methods such as a liquid-phase process, light initiated processes, or the use of fluidized beds with a catalyst have been proposed for increased selectivity, although it is not known if these are commonly used in industry today (Holbrook 2003).

The nationally aggregated production volume reported to the EPA Chemical Data Reporting (CDR) database for 2019 was between 250 and 500 million pounds (EPA 2023a). This quantity has remained consistent between 2016 and 2019. Four companies reported manufacturing chloroform at a total of six sites to the 2020 CDR database, covering the years 2016–2019 (EPA 2023a). These include Shintech Louisiana LLC (Plaquemine, Louisiana), EMD Holding Corporation (Rockland, Massachusetts); Olin Corporation (Freeport, Texas), and Occidental Petroleum Corporation (Wichita, Kansas; Geismar, Louisiana; and La Porte, Texas). This may not be an exhaustive list of producers; companies must meet a volume threshold to trigger reporting to the EPA CDR database, and some companies' site activities were not available in the public dataset.

Table 5-1 summarizes information on companies that reported the production, import, or use of chloroform for the Toxics Release Inventory (TRI) in 2022 (TRI22 2024). TRI data should be used with caution since only certain types of industrial facilities are required to report. This is not an exhaustive list.

Table 5-1. Facilities that Produce, Process, or Use Chloroform

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AL	5	0	9,999,999	1, 5, 6, 10
AR	1	10,000	99,999	1, 2, 3, 5, 9, 12
CO	1	100,000	999,999	6, 10, 14
CT	1	10,000	99,999	9
FL	1	100	999	1, 5
IL	3	10,000	99,999	1, 5, 10, 12
IN	1	1,000	9,999	9, 12
KS	2	100	9,999,999	1, 4, 12

Table 5-1. Facilities that Produce, Process, or Use Chloroform

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
KY	3	10,000	9,999,999	1, 2, 3, 4, 6, 9
LA	12	0	99,999,999	1, 3, 4, 5, 6, 9, 12, 13
MA	2	10,000	99,999	10, 11
MI	1	100,000	999,999	7, 9, 12
MO	4	0	999,999	1, 5, 7, 9, 10, 12
MT	1	10,000	99,999	10
NC	2	100	99,999	1, 5, 12
NE	1	10,000	99,999	9, 12
NJ	2	1,000	999,999	9, 10
NY	1	10,000	99,999	12
OH	9	100	999,999	7, 8, 9, 12
OR	1	1,000	9,999	10
PA	1	10,000	99,999	1, 5
SC	3	100	9,999	1, 5, 12
TX	14	0	49,999,999	1, 2, 3, 4, 5, 6, 9, 10, 12, 13, 14
UT	2	100	99,999	1, 5, 9, 10, 12
VA	1	1,000,000	9,999,999	10
WI	1	10,000	99,999	7, 9, 10
WV	2	1,000	99,999	1, 6, 10, 13

[a] Post office state abbreviations used.
[b] Amounts on site reported by facilities in each state.
[c] Activities/uses:

1. Produce
2. Import
3. Used Processing
4. Sale/Distribution
5. Byproduct
6. Reactant
7. Formulation Component
8. Article Component
9. Repackaging
10. Chemical Processing Aid
11. Manufacture Aid
12. Ancillary
13. Manufacture Impurity
14. Process Impurity

Source: TRI22 2024 (Data are from 2022)

5.2.2 Import/Export

Three companies reported importing chloroform to the 2020 CDR database between 2016 and 2019: ICC Industries, Inc., EMD Holding Corporation, and INEOS Chlor Americas Inc (EPA 2023a). Import volumes were not reported. No companies reported exportation of chloroform. This may not be complete information, as volume thresholds must be met to trigger reporting to the CDR database, and some data were not available in the public dataset. The U.S. International Trade Commission reported a total import volume of 357,517 kg for chloroform in 2022 (USITC 2023).

5.2.3 Use

The major use for chloroform is in the manufacture of the refrigerant HCFC-22, also known as R-22 (Ohligschläger et al. 2019). Despite the phase out of HCFC-22 as a refrigerant, chloroform demand has remained stable due to the use of chlorodifluoromethane as a precursor to feedstocks for fluoropolymers, such as polytetrafluoroethylene. These polymers are used as corrosion-resistant liners in steel pipes and reactors for the chemical and pharmaceutical industry and in electronics and medical equipment. Additionally, the polymers are used as coatings for nonstick cookware and waterproof, breathable fabrics. The polymers are also used as lubricators for sprays and greases and are in specialty roofing and gliding materials (Ohligschläger et al. 2019). Chemical manufacturing companies reported industrial usage of chloroform as an intermediate, solvent, and laboratory chemical in chemical manufacturing (EPA 2023a). There were no details on consumer product usage reported (EPA 2023a).

Chloroform has been used in the past as a solvent or an extraction solvent for fats, oils, greases, resins, lacquers, rubber, alkaloids, gums, waxes, gutta-percha, penicillin, vitamins, flavors, floor polishes, and adhesives in artificial silk manufacture. It has also been used as a dry-cleaning spot remover, in fire extinguishers, and as an intermediate in the manufacture of dyes and pesticides (Deshon 1979). Chloroform has been used as a fumigant and insecticide (Holbrook 2003); however, there are no currently active pesticide products containing chloroform as an active or inert ingredient (EPA 2023b, 2023c).

Chloroform was previously used as an anesthetic, but it has been replaced by safer and more versatile materials (Deshon 1979). The U.S. Food and Drug Administration (FDA) banned chloroform use in drug, cosmetic, and food packaging products in 1976 (IARC 1979). However, since the ban did not include drug products that contain chloroform in residual amounts, it may still be used as a solvent in the pharmaceutical industry and may be present as a byproduct from the synthesis of drug ingredients (IARC 1979). Federal regulations for indirect food additives currently allow chloroform use as a processing aid in polymers, adhesives, and as components of coatings (FDA 2024a, 2024b, 2024c, 2024d). In the early 1990s, chloroform was still reportedly used as a local anesthetic and solvent in certain dental endodontic (gutta-percha root canal) surgery procedures and in topically applied aspirin-chloroform mixtures for pain relief in severe cases of herpes zoster (shingles) or post-therapeutic neuralgia (King 1993; McDonald and Vire 1992); however, it is uncertain if these uses still occur today.

5.2.4 Disposal

Chloroform has been identified as a hazardous waste by EPA, and disposal of this waste is regulated under the Federal Resource Conservation and Recovery Act (RCRA) (EPA 1988a, 1989). Specific information regarding federal regulations on chloroform disposal on land is available in the Code of Federal Regulations (EPA 1988a, 1989). Ultimate disposal of chloroform, preferably mixed with another combustible fuel, can be accomplished by controlled incineration. Complete combustion must be ensured to prevent phosgene formation, and an acid scrubber should be used to remove the haloacids produced. Chloroform may also be disposed of by liquid injection incineration, although the use of this method has not been verified. Chloroform has been previously used in some pesticides, so the disposal of old pesticide containers may be relevant. Combustible containers from organic or many metallo-organic pesticides could be disposed of in pesticide incinerators or in specified landfill sites. Except for the TRI statistics, no data were located regarding the approximate amounts of chloroform disposed or released to environmental media.

Wastewater treatment methods that limit chloroform formation as a disinfection byproduct have been developed. In a study comparing disinfection byproduct formation following wastewater treatment by ultraviolent irradiation (UV), vacuum ultraviolet radiation (VUV), and chlorination, products were limited when under 40 mJ/cm^2 UV and 3 mg/L chlorine were applied at the same time (Du et al. 2024). Applying VUV at the same time as the UV/chlorine treatment resulted in accelerated disinfection with less formation of chloroform. Conversely, when these treatment methods were applied sequentially, an increase in formation of disinfection byproducts, including chloroform, was observed. In another study, UV/peroxymonosulfate treatment was the most effecting post treatment in reducing chloroform formation, followed by UV and UV/H_2O_2 (Huang et al. 2022).

5.3 RELEASES TO THE ENVIRONMENT

The Toxics Release Inventory (TRI) data should be used with caution because only certain types of facilities are required to report (EPA 2022). This is not an exhaustive list. Manufacturing and processing facilities are required to report information to the TRI only if they employ ≥10 full-time employees; if their facility's North American Industry Classification System (NAICS) codes is covered under EPCRA Section 313 or is a federal facility; and if their facility manufactures (defined to include importing) or processes any TRI chemical in excess of 25,000 pounds, or otherwise uses any TRI chemical in excess of 10,000 pounds, in a calendar year (EPA 2022).

5.3.1 Air

Estimated releases of 258,677 pounds (~117 metric tons) of chloroform to the atmosphere from 78 domestic manufacturing and processing facilities in 2022, accounted for about 59% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). These releases are summarized in Table 5-2.

Table 5-2. Releases to the Environment from Facilities that Produce, Process, or Use Chloroform[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	5	4,524	10	1,134	0	0	4,534	1,134	5,668
AR	1	3	0	0	28	0	3	28	31
CO	1	85	0	0	0	0	85	0	85
CT	1	0	0	0	0	0	0	0	0
FL	1	31,400	1,280	0	42	0	32,722	0	32,722
IL	3	1,280	5	0	14	0	1,280	19	1,299
IN	1	169	0	0	0	0	169	0	169
KS	2	4,153	0	12,137	0	0	16,290	0	16,290
KY	3	829	56	0	0	0	834	52	886
LA	12	85,215	2,456	0	142	391	87,671	533	88,203
MA	2	833	1	0	0	88,894	833	88,895	89,728
MI	1	124	1	0	0	0	124	1	125
MO	4	2,199	3,605	0	0	1,345	5,799	1,350	7,149
MT	1	1,901	3	0	0	0	1,901	3	1,904
NE	1	252	0	0	0	0	252	0	252
NJ	2	69	55	0	0	0	69	55	124
NY	1	174	0	0	0	0	174	0	174
NC	2	36,171	15	0	37	0	36,223	0	36,223
OH	9	665	159	0	207	68,987	810	69,208	70,018
OR	1	38	0	0	0	0	38	0	38
PA	1	4,716	1	0	0	0	4,716	1	4,717
SC	3	6,183	12	0	0	0	6,195	0	6,195
TX	14	67,265	675	2	28	37	67,960	46	68,006
UT	2	0	0	0	14	0	0	14	14
VA	1	10,094	27	0	0	0	10,121	0	10,121
WV	2	255	0	0	0	0	255	0	255

Table 5-2. Releases to the Environment from Facilities that Produce, Process, or Use Chloroform[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
WI	1	80	34	0	0	0	80	34	114
Total	78	258,677	8,395	13,273	511	159,654	279,138	161,372	440,510

[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, wastewater 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: TRI22 2024 (Data are from 2022)

EPA’s National Emission Inventory (NEI) database contains information regarding sources that emit criteria air pollutants (CAPs) and their precursors, and hazardous air pollutants (HAPs) for the 50 United States, Washington DC, Puerto Rico, and the U.S. Virgin Islands. Emissions are estimated 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. Chloroform emissions estimated from the 2017 inventory are summarized in Table 5-3.

Table 5-3. National Emission Inventory (NEI) Total National Emissions for Chloroform Estimated by Sector, 2020

Sector	Emissions (pounds)
Industrial processes; pulp and paper	355,872
Industrial processes; cement manufacturing	254,170
Industrial processes; chemical manufacturing	208,313
Agriculture; livestock waste	203,872

Table 5-3. National Emission Inventory (NEI) Total National Emissions for Chloroform Estimated by Sector, 2020

Sector	Emissions (pounds)
Industrial Processes; NEC	174,915
Waste disposal	133,052
Industrial processes; petroleum refineries	52,928
Industrial processes; oil and gas production	27,518
Fuel combustion; industrial boilers, ICEs; natural gas	11,315
Solvent; industrial surface coating and solvent use	10,098
Fuel combustion; industrial boilers, ICEs; biomass	9,523
Industrial processes; storage and transfer	8,499
Fuel combustion; comm/institutional; biomass	6,033
Fuel combustion; electric generation; coal	5,749
Fuel combustion; electric generation; biomass	1,568
Fuel combustion; industrial boilers, ICEs; coal	722
Industrial processes; non-ferrous metals	402
Solvent; graphic arts	320
Fuel combustion; electric generation; natural gas	288
Fuel combustion; commercial/institutional; natural gas	266
Solvent; dry cleaning	261
Fuel combustion; industrial boilers, ICEs; oil	252
Industrial processes; ferrous metals	203
Fuel combustion; industrial boilers, ICEs; other	201
Fuel combustion; electric generation; other	122
Fuel combustion; commercial/institutional; other	116
Industrial processes; mining	110
Solvent; degreasing	66
Fuel combustion; commercial/institutional; coal	10
Bulk gasoline terminals	0
Fuel combustion; comm/institutional; oil	0
Fuel combustion; residential; other	0
Fuel combustion; electric generation; oil	0

ICE = internal combustion engine; NEC = not elsewhere classified

Source: EPA 2023f

Direct releases to the atmosphere are expected to occur during the manufacture, loading, and transport of chloroform (EPA 1985a, 1985b). The current largest estimated anthropogenic sources of atmospheric emissions are from the manufacturing of paper and pulp, cement, and chemicals (EPA 2023f). Indirect chloroform releases have resulted from its use in the manufacture of HCFC-22, fluoropolymers, pharmaceuticals, ethylene dichloride, dyes, and fumigants (Deshon 1979; EPA 1985a, 1985b; Holbrook 2003). By one estimate for the state of Minnesota, point sources were estimated to be the greatest contribution to chloroform emissions (~80%) (Pratt et al. 2000). Global emissions of chloroform were estimated to be between 260 and 400 giga grams (~570–880 million pounds) per year for 2016, an estimated 20% increase from 2011 (WMO 2018).

Chloroform releases can also result from its formation and subsequent volatilization from chlorinated waters including drinking water, domestic water use, municipal and industrial wastewaters, process waters and effluent from the bleaching of pulp in pulp and paper mills, cooling tower water, and swimming pool water, and whirlpool spa water (Benoit and Jackson 1987; EPA 1985a, 1985b; Hoigne and Bader 1988; Scott et al. 2020; Shepherd et al. 1996). Volatilization of chloroform formed during wastewater treatment has been estimated to be contributing 55,000 tons/year to the atmosphere (Ohligschläger et al. 2019). Ranges of 1.27–155 μg/m^3 chloroform were detected in samples collected in the headspace of primary sedimentation and secondary treatment tanks at an industrial wastewater treatment plant in Spain (Ramírez et al. 2011). Previous estimates of chloroform released were 183 mg/person/year from showering, and 120–140 mg chloroform/person/year from laundry loads (Shepherd et al. 1996). Domestic water usage is expected to be the main source of chloroform emissions to indoor air. Increased release rates of the chloroform in water can be expected from chloroform-containing water that is heated (e.g., water used for cooking, showers, swimming pools, and spas). Aeration and use of groundwater contaminated with chloroform are also potential sources of emission to the atmosphere (Crume et al. 1990).

Chloroform is released as a result of hazardous and municipal waste treatment processes. The chloroform released may have initially been present in the waste or possibly formed during chlorination treatment (Corsi et al. 1987; EPA 1990b; Namkung and Rittmann 1987). Releases may also occur from hazardous waste sites and sanitary landfills where chloroform was disposed, and from municipal and hazardous waste incinerators that burn chloroform-containing wastes or produce chloroform during the combustion process (LaRegina et al. 1986; Travis et al. 1986).

In the past, minor releases may have resulted from the use of consumer products (e.g., certain air deodorizers and cleaning products) that contained chloroform as a component or residual product (Bayer et al. 1988; Wallace et al. 1987a). Chloroform is widely used in laboratory work as an extractant, and the deuterated form of chloroform is used as a solvent in nuclear magnetic resonance spectroscopy.

Some studies estimated that only 9.5–10% of chloroform released to the atmosphere is anthropogenic (Laturnus et al. 2002; USGS 2004); however, others suggested that up to 50% of the total global emission is attributable to man-made sources (Sekar et al. 2022). Natural sources of chloroform include volcanic emissions and biomass burnings, and fluxes of chloroform to the atmosphere have been measured from marine and terrestrial environments, such as tropical oceans, forest soil, rice fields, and peatland. Previously estimated emissions for these sources were 360,000 tons/year from oceans, 220,000 tons/year from soil, and 15 tons/year from other natural processes (Ohligschläger et al. 2019).

5.3.2 Water

Estimated releases of 8,395 pounds (~3.8 metric tons) of chloroform to surface water from 78 domestic manufacturing and processing facilities in 2022, accounted for 2% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). This estimate includes releases to wastewater treatment and publicly owned treatment works (POTWs) (TRI22 2024). These releases are summarized in Table 5-2.

Other than TRI data, current, more comprehensive quantitative data or estimates of chloroform releases to natural waters are lacking. Disinfection of water supplies by chlorination is nearly universal; thus, residual chloroform is found in most drinking water as a disinfection byproduct. This widespread chloroform contamination makes direct releases challenging to measure.

Chlorination of municipal and industrial wastewaters at wastewater treatment plants, process waters, and effluent from the bleaching of pulp in pulp and paper mills, cooling-tower water, and swimming-pool and whirlpool-spa water will also result in chloroform formation (Benoit and Jackson 1987; Comba et al. 1994; EPA 1985a, 1985b, 1990a; Hoigne and Bader 1988). Maximum chloroform formation under simulated chlorination treatment of raw water was 11–13 mg/L (Chaidou et al. 1999). The use of modern treatment facilities may reduce the amounts of chloroform released to environmental waters. This has been demonstrated at a modern kraft pulp mill (Paasivirta et al. 1988); however, much of the chloroform removed from the wastewater may be released to the atmosphere by volatilization. Release of chloroform to groundwater has resulted from improper disposal of chloroform-containing waste at hazardous waste sites (Clark et al. 1982; Dewalle and Chian 1981; Harris et al. 1984; Sawhney 1989).

Microplastics in wastewater have been shown to contribute to disinfection byproduct formation. As the microplastics degrade under UV or natural sunlight, dissolved organic matter (DOM) is released, which is used as substrate for chloroform formation during chlorination (Chen et al. 2024). Chloroform formation potentials determined for DOM leached from polystyrene and polypropylene microplastics under simulated natural water and chlorination conditions were 60.3±7.8 and 73.7±9.8 μg/g respectively (Yan et al. 2024).

An additional minor source of water contamination may be atmospheric rainout since chloroform has been found in rainwater (Kawamura and Kaplan 1983). Chloroform has been detected in urban stormwater (Lopes and Bender 1998). Other sources of chloroform release to surface water include breweries and thermal combustion of plastics (EPA 1985a).

Direct releases to water are expected via wastewaters generated during chloroform manufacture and its use in the manufacture of other chemicals and materials (EPA 1985a). Direct discharge sources are expected to be relatively minor contributors to total chloroform emissions to water relative to the formation of chloroform resulting from the chlorination of drinking water or chlorination to eliminate pathogens in discharged wastes or other process waters (EPA 1985a).

Natural formation of chloroform in water occurs through abiotic and biotic processes, most of which are reported in marine environments (EPA 1985a; Laturnus et al. 2002). One estimate reported production of 350 giga gram (~770 million pounds) per year in the ocean (USGS 2004). Due to its volatility, chloroform formed in surface water is expected to emit to the atmosphere. Chloroform is also found in groundwater that originates from its natural production in soil (Gron et al. 2012) and is persistent in environments where oxygen is present (Hunkeler et al. 2012).

5.3.3 Soil

Estimated releases of 551 pounds (~0.23 metric tons) of chloroform to soil from 87 domestic manufacturing and processing facilities in 2022, accounted for about <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). An additional 13,273 pounds (~6.0 metric tons), accounted for about 3% of the total environmental emissions, were released via underground injection (TRI22 2024). These releases are summarized in Table 5-2.

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

Chloroform is produced in terrestrial environments by biomediated processes; one of the more significant sources seems to be forest soil, although fluxes from grasslands, dry swamplands, and peat moorland have also been detected (Hoekstra et al. 2001; Laturnus et al. 2002). Chloroform has also been detected in the emissions of aerobic composting of vegetable waste and cow manure (Qu et al. 2022). Chloroform emissions from aerobic composting varied temporally and ranged from not detected (detection limit to reported) to 20 $\mu g/m^3$. However, chloroform formed in terrestrial environments may not remain there; it is expected to volatilize to the atmosphere or migrate to groundwater. The volatilization flux of chloroform from the soil of a Douglas fir forest was 1,000 ng/m^2/hour (Hoekstra et al. 2001). Fluxes from rice fields have been measured in the range of 1.4×10^4–9.6×10^4 ng/m^2/day (Laturnus et al. 2002). Fluxes from termite mounds have also been detected at 0.0002 ng/cm^2/second (Laturnus et al. 2002).

5.4 ENVIRONMENTAL FATE

5.4.1 Transport and Partitioning

Air. Based on vapor pressures of 159–197 mm Hg at 20–25°C, chloroform is expected to exist almost entirely in the vapor phase in the atmosphere (Boublik et al. 1984; Eisenreich et al. 1981). Large amounts of chloroform in the atmosphere may be removed by wet deposition since chloroform has significant solubility in water (Table 4-2). This is confirmed by its detection in rainwater (Kawamura and Kaplan 1983). Most chloroform is removed from the atmosphere in precipitation and is likely to re-enter the atmosphere by volatilization. Since chloroform has a relatively long half-life in the atmosphere, long-range transport is possible. Trace amounts of chloroform have been documented in air samples from remote, often relatively pristine, areas of the world (Class and Ballschmidter 1986). This may also be due to natural chloroform formation via reaction of naturally generated chlorinated oxidants with organic matter (Laturnus et al. 2000; Laturnus et al. 2002).

Water. Based on the measured Henry's law constant of 3.00–3.67x10^{-3} atm-m^3/mol, the dominant fate process for chloroform in surface waters is volatilization (Gossett 1987; Nicholson et al. 1984). Chloroform present in surface water is expected to volatilize rapidly to the atmosphere. An experimental half-life of 18–25 minutes has been measured for volatilization of chloroform from a 1-ppm solution with a depth of 6.5 cm that was stirred with a shallow pitch propeller at 200 rotations per minute at 25°C under still air (≈0.2 mph air currents) (Dilling 1977; Dilling et al. 1975). Using the Henry's law constant, a half-life of 3.5 hours was calculated for volatilization from a model river that is 1 meter deep flowing at 1 m/second, with a wind velocity of 3 m/second, and neglecting adsorption to sediment (Lyman et al. 1982).

Sediment and Soil. In soil, the dominant transport mechanism for chloroform near the surface will likely be volatilization. A chloroform soil-air partition coefficient (K_{SA}) of 43.8 in soil with 67% water content at 25 °C was estimated based on the air-octanol partition coefficient (log K_{oa}) of 2.79 L/kg (Ahn et al. 2020). Volatilization rates were comparable over a wide variety of soil types and were not concentration dependent (Park et al. 1988). However, volatilization may be impacted by flow rate in soil. In soil column studies using fine sandy soil, 75% of the chloroform initially present in water volatilized when applied at a slow flow rate compared to 54% volatilization when applied at a faster flow rate (Piwoni et al. 1986; Wilson et al. 1981). All, or nearly all, of the remaining chloroform traveled through the soil because of its low adsorption onto soil. Another laboratory study of 15 common volatile or semi-volatile organic chemicals reported a half-life for chloroform of 4.1 days, which assumed first-order kinetic decay (Anderson et al. 1991).

The leaching potential of chloroform is confirmed by the detection of chloroform in groundwater, especially at hazardous waste sites (Clark et al. 1982; Dewalle and Chian 1981; Harris et al. 1984; Hunkeler et al. 2012; Sawhney 1989). Measured log K_{oc} values of 1.5–2.4 support the low sorption observed in laboratory studies (Sabljic 1984). Little or no chloroform concentration was observed on peat moss, clay, dolomite limestone, or sand added to water (Dilling et al. 1975). Chloroform slightly adsorbed to aquifer solids in laboratory studies utilizing different amounts of two different aquifer materials, with K_{oc} values ranging from 63.4 to 398 (log K_{oc}=1.80–2.59). The study authors reported higher adsorption with increasing organic content of the solids (Uchrin and Mangels 1986). Another study measured K_{oc} values ranging from 45 to 80 (log K_{oc}=1.65–190) in soil (Sabljic 1984; Wilson et al. 1981).

Other Media. Chloroform does not appear to bioconcentrate in higher aquatic organisms, based upon measured bioconcentration factors (BCF) of 6 and 8 for bluegill sunfish (*Lepomis macrochirus*) (Barrows et al. 1980; Veith et al. 1980) and of 13 for the common carp (*Cyprinus carpio*) (NITE 1980). A BCF of 690 experimentally determined in green algae, *Selenastrum capricornutum*, suggests that the compound has some tendency to concentrate in nonvascular aquatic plants (Mailhot 1987). No data regarding the biomagnification potential of chloroform were found. Based upon the observed BCF, however, significant biomagnification of chloroform is apparently unlikely.

5.4.2 Transformation and Degradation

Air. The vapor-phase reaction of chloroform with photochemically generated hydroxyl radicals is the dominant degradation process in the atmosphere. The rate constant for this process at 25°C has been experimentally determined as 1.05×10^{-13} cm^3/molecule-second, which corresponds to a half-life of ≈102 days based upon a 12-hour sunlit day in a typical atmosphere containing 1.5×10^6 hydroxyl radicals/cm^3 (DOT 1980; Singh et al. 1981). Chlorinated degradation products from reaction with hydroxyl radicals include inorganic chlorine, hydrogen chloride, formyl chloride, and phosgene (Holbrook 2003; Tsai 2017).

Chloroform is more reactive in photochemical smog conditions, where the approximate half-life is 11 days (Dimitriades and Joshi 1977). Direct photolysis of chloroform will not be a significant degradation process in the atmosphere. Chloroform solutions sealed in quartz tubes and exposed to sunlight for 1 year degraded at almost the same rate as solutions in sealed tubes stored in the dark, indicating that little or no photodegradation of the compound had occurred (Dilling et al. 1975). This is expected because chloroform does not show significant light absorbance at wavelengths >290 nm (Hubrich and Stuhl 1980).

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

1987). This latter process is probably too slow to effectively compete with volatilization as a removal process from surface waters. Under iron- or sulfate-reducing conditions, chloroform can be reduced to dichloromethane (USGS 2004).

Biological degradation of chloroform has been studied primarily under conditions of batch process operations at wastewater treatment plants or as a remediation option at hazardous waste disposal sites. Above certain dosage levels, chloroform becomes toxic to anaerobic and aerobic microorganisms. This is especially noticeable for biological treatment facilities that use anaerobic digestion systems, where sustained inputs with chloroform concentrations approaching 100 mg/L can all but eliminate methanogenic (methane-fermenting) bacteria (Rhee and Speece 1992). Other studies have shown appreciable inhibition of methanogenesis, with levels of chloroform of 1 mg/L (Hickey et al. 1987). Other chlorinated hydrocarbons, and particularly such common 2-carbon chlorinated aliphatics as TCE, can similarly inhibit bacteria found in sewage sludges (Long et al. 1993; Rhee and Speece 1992). Similar inhibition effects can be the result of heavy metal toxics, zinc being particularly stressful to methanogenic bacteria (van Beelen et al. 1994; van Vlaardingen and van Beelen 1992).

Studies of natural waters or wastewaters, where it is difficult to control the levels of specific chemicals or preclude inputs of other toxicants, yield a wide variety of results on the efficiencies of chloroform biodegradation. For instance, little or no degradation was observed during 25 weeks in aqueous aerobic screening tests utilizing primary sewage effluent inocula (Bouwer et al. 1981a), or in 2 weeks following a standard readily biodegradability screening test (Organization for Economic Co-operation and Development [OECD] 301C) in aerobic inoculum (NITE 2010). No chloroform degradation was observed in aerobic biofilm column studies (Bouwer et al. 1981b). Aerobic screening tests utilizing settled domestic wastewater as inoculum reported significant loss of chloroform, 46–49% loss in 7 days, indicating degradation; however, at least some of the loss was apparently attributable to volatilization (Tabak et al. 1981).

Under the proper conditions, chloroform appears to be much more susceptible to anaerobic biodegradation. Bouwer et al. (1981a) determined that degradation of chloroform under anaerobic conditions was more rapid at lower chloroform concentrations (81 and 99% degradation after 2 and 16 weeks, respectively, at 16 ppb) compared to higher concentrations (78% degradation after 16 weeks, at 157 ppb). Reported anaerobic degradation products were dichloromethane and carbon dioxide (Vickstrom et al. 2017). No degradation was observed when chloroform was incubated with aquifer material under anaerobic conditions for 27 weeks (Wilson et al. 1981). Several studies have indicated that in the presence of acid, chloroform will more easily undergo anaerobic degradation (Gupta et al. 1996). Up to 96% chloroform removal was achieved with concentrations up to 16.74 μM (2,000 ppb) with acetic acid as the primary substrate. Dichloromethane was identified as the primary transformation product via reductive dehalogenation.

In the absence of toxicity from other solvents, chlorinated hydrocarbons, or heavy metals, and where chloroform concentrations can be held below approximately 100 ppb, both aerobic and anaerobic bacteria can biodegrade chloroform, with removal rates well over 80% in a period of 10 days (Long et al. 1993). Deviations from these ideal conditions can lead to lower removal efficiencies. These biodegradation reactions generally lead to the mineralization of the chloroform to chlorides and carbon dioxide (Bouwer and McCarty 1983; Rhee and Speece 1992). One study, however, documented the production of the toxicant methylene chloride (dichloromethane) from the breakdown of chloroform containing wastes in a mixed culture of bacteria from sewage sludge (Rhee and Speece 1992, citing results from work at Tyndall Air Force Base, Florida). However, caution should be exercised in generalizing without site-specific evidence since commercial grades of chloroform will often contain methylene chloride as an impurity. In waters containing mixtures of different chlorinated aliphatics, biodegradation may produce new chloroform, at least as a temporary byproduct, the breakdown of carbon tetrachloride into chloroform having been confirmed in laboratory studies (de Best et al. 1998; Long et al. 1993; Picardal et al. 1993).

Sediment and Soil. Little information was located regarding the degradation of chloroform in soil. Based on data for degradation in water, chemical degradation in soil is not expected to be significant. Under proper redox (iron- or sulfate-reducing) conditions, chloroform can be abiotically reduced to dichloromethane (USGS 2004). The available soil data suggest that chloroform biodegradation rates in soil may vary, depending upon conditions.

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

Other Media. No studies on the transformation and degradation of chloroform in biological or other systems were located.

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to chloroform depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of chloroform 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 chloroform levels monitored or estimated in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable.

Table 5-4 shows the reported lowest limit of detections that are achieved by standard analytical analysis in environmental media. An overview summary of the range of concentrations detected in environmental media is presented in Table 5-5.

Table 5-4. Lowest Limit of Detection Based on Standards[a]

Media	Detection limit	Reference
Air (ppbv)	0.020 0.01–1.8 300	WHO 2004 EPA 2023d NIOSH 2018
Drinking water (ppb)	0.001 0.055	WHO 2004 EPA 1995
Surface water and groundwater (ppb)	0.001 1 30–900	WHO 2004 EPA 2014 EPA 2018a
Soil (ppb)	0.0015 1 30–900	Laturnus et al. 2000 EPA 2014 EPA 2018a

Table 5-4. Lowest Limit of Detection Based on Standards[a]

Media	Detection limit	Reference
Whole blood (ppb)	0.0021–0.008	CDC 2022a, 2022b

[a] Detection limits based on using appropriate preparation and analytics. These limits may not be possible in all situations.

Table 5-5. Summary of Environmental Levels of Chloroform

Media	Low	High	For more information
Outdoor air (ppbv)	0.008	27.2	Section 5.5.1
Indoor air (ppbv)	0.04	93.60	Section 5.5.1
Surface water (ppb)	0.26	85	Section 5.5.2
Groundwater (ppb)	<0.2	324	Section 5.5.2
Groundwater (ppb)	<0.2	120	Section 5.5.2
Drinking water (ppb)	<0.2	75	Section 5.5.2
Sediment (ppb)	0.0017	539	Section 5.5.3
Soil (ppb)	–	–	Section 5.5.3
Food (ppb)	0	176	Section 5.5.4

Detections of chloroform in air, water, and soil at NPL sites are summarized in Table 5-6.

Table 5-6. Chloroform Levels in Water, Soil, and Air of National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
Water (ppb)	22	29.9	17.0	474	274
Soil (ppb)	77	250	57.1	115	82
Air (ppbv)	0.881	1.32	28.9	78	56

[a] Concentrations found in ATSDR site documents from 1981 to 2022 for 1,868 NPL sites (ATSDR 2022c). Maximum concentrations were abstracted for types of environmental media for which exposure is likely. Pathways do not necessarily involve exposure or levels of concern.

5.5.1 Air

While chloroform is found abundantly in the environment due to both natural and anthropogenic sources, it is infrequently monitored (Sekar et al. 2022). Fairly stable global averages of 7.3–7.7 parts per trillion by volume (pptv) chloroform in air were reported based on measurements between 1997 and 2010; an increase to 8.9 pptv was reported in 2016, likely based on increased anthropogenic inputs (WMO 2018). The Air Quality System (AQS) reports the average ambient chloroform concentrations in air from hazardous air pollutant monitors across the United States; levels from 2018 to 2022 have remained relatively constant between 0.018 and 0.04 ppbv (EPA 2023d). The data are summarized in Table 5-7. In an assessment of 3,650 urban and rural locations in Minnesota, chloroform was detected (>0.023 ppbv) in 1,445 samples, the mean concentration was 0.027 ppbv, with a maximum of 1.41 ppbv (Pratt et al. 2000).

Table 5-7. Summary of Annual Concentrations of Chloroform (ppbv) Measured in Ambient Air at Locations Across the United States[a,b]

Year	Number of monitoring locations	Number of samples	Average	Maximum
2018	199	9,900	0.018	9.1
2019	129	7,053	0.025	27.2
2020	166	8,461	0.04	8.8
2021	170	11,466	0.03	7.7
2022	125	2,988	0.032	10.8
2023[c]	115	2,443	0.025	5.5

[a] Values were originally reported in parts per billion carbon (ppbC) and converted to ppbv.
[b] 24-hour sampling period.
[c] As of October 26, 2023.

Source: EPA 2023d

Chloroform levels in air can be much higher in areas near hazardous waste sites (Stephens et al. 1986). The median concentration for source-dominated areas in the United States was 0.82 ppbv for data reported between 1977 and 1980, and 0.51 ppbv for data reported in 1987 (EPA 1982, 1988b). Certain source-dominated areas contained much higher chloroform levels. The ambient air concentrations outside homes in Love Canal, New York, in 1978, were 2–22 ppbv, and the maximum concentration found in ambient air at 20 California municipal landfills was 610 ppbv (Barkley et al. 1980; Wood and Porter 1987). Concentrations of 0.29–6 ppbv were found in air samples taken from five hazardous waste sites in New Jersey (LaRegina et al. 1986). Ambient air samples measured near a hazardous waste landfill contained ≤1 ppbv chloroform. Other source-dominated areas that may have ambient air chloroform concentrations significantly higher than background levels include areas near facilities that treat hazardous and municipal waste, as well as areas near contaminated groundwater, and municipal and hazardous waste incinerators (Corsi et al. 1987; EPA 1990a; LaRegina et al. 1986; Namkung and Rittmann 1987; Travis et al. 1986).

Elevated chloroform concentrations in air are present near some industrial sites, such as paper mills. Air samples were collected from 2017 to 2018 from the Lewis-Clark Valley in Washington, where a pulp paper mill operates (Scott et al. 2020). Annual chloroform averages were 0.15±0.19 $\mu g/m^3$ (range: 0.02–1.55 $\mu g/m^3$) for 2017 and 0.26±0.34 $\mu g/m^3$ (range: 0.03–3.33) for 2018.

The EPA (2016) includes chloroform in its Vapor Intrusion Screening Levels (VISL) Calculator, indicating that it is sufficiently volatile and sufficiently toxic to be considered a concern for vapor intrusion from soil water and groundwater. Accordingly, ATSDR (2016) recommends that health assessors should evaluate potential health implications of vapor intrusion for chloroform during site risk assessments.

A review of ATSDR public health assessments completed between 1994 and 2009 identified 33 sites with chloroform detected in soil gas, crawl space, indoor air, or outdoor air (Burk and Zarus 2013). Indoor air was sampled at 15 of the sites with chloroform detected from 0.03 to 23 $\mu g/m^3$ (0.01–4.7 ppbv). Chloroform was detected in soil gas at the 15 sites ranging from 2.4 to 146 $\mu g/m^3$ (0.24–29 ppbv) and in outdoor air ranging from 0.29 to 2.2 $\mu g/m^3$ (0.06–0.45 ppbv).

Data from the EPA vapor intrusion database found that chloroform was detected in 68.5% of 2,278 indoor air samples collected from 1990 to 2005 (EPA 2012). The maximum concentration of chloroform detected in indoor air at residential sites included in the vapor intrusion database was reported as 1.4 $\mu g/m^3$ (0.31 ppbv) (EPA 2012). In a study of chloroform vapor intrusion, the source was hypothesized to be chlorinated water in sewer lines below the impacted residence (McHugh et al. 2017).

One of the most significant indoor sources of chloroform is chlorinated tap water, and taking showers is expected to contribute a substantial amount to the indoor chloroform levels (Andelman 1985a, 1985b; EPA 1987; Kerger et al. 2005; Wallace 1997). In 100 residences monitored in suburban and rural areas of New Jersey, concentrations of chloroform detected in indoor air ranged from 0.20 to 1.2 ppbv (Weisel et al. 2008). Only 29% of samples were above the detection limit; however, the high limit of detection for about half of the samples (0.49 ppbv for 48 samples and 0.20 ppbv for 52 samples) limits the usefulness of this study. In portable classrooms used in kindergarten through grade 12 public schools in Los Angeles, California, daily average chloroform concentrations ranged from 0.02 to 0.06 ppbv (Shendell et al. 2004). Median indoor chloroform levels were 0.39 ppbv in 99 private residences monitored in southeast Louisiana between 2013 and 2015 (Wickliffe et al. 2020). Chloroform was not detected in the main school building classroom during June but was detected between <0.027 and 0.08 ppbv during the winter and fall months.

The air around swimming pools may also contain chloroform. This is especially likely in heated, indoor pools, which can approximate the conditions found in shower stalls. Concentrations ranging from 3.99 to 205.6 ppbv have been measured in air at indoor swimming pools (Ahmadpour et al. 2022; Font-Ribera et al. 2010; Nitter and Svendsen 2019; Sa et al. 2011). In one study of four Canadian indoor swimming pools, there was not a significant difference between chloroform concentrations in air samples collected at 0.5 or 1.5 m above the water surface (Ahmadpour et al. 2022). Temporal variation in chloroform concentrations led the study authors to recommend an 8-hour sampling strategy instead of using 2-hour samples for assessing worker chloroform exposures (Ahmadpour et al. 2022).

Due to chloroform's potential to be transported long distances in air and production in the environment, chloroform has been detected in many remote locations. In 2003, the concentration of chloroform in air in remote areas in the United States ranged from 0.008 to 0.0098 ppbv (McCarthy et al. 2006). It is noted that these reported concentrations are below the reported limits of detection for standard analytical methods shown in Table 5-4. McCarthy et al. (2006) does not report the limit of detection for the analytical method used in their analysis.

5.5.2 Water

The EPA maintains a Water Quality Portal (WQP) database, which aggregates environmental monitoring data from the National Water Information System (NWIS) and STORage and RETrieval (STORET) systems. A summary of the data for ambient surface and groundwater from recent years is provided in Table 5-8 (WQP 2024). Chloroform has been detected in surface water and groundwater. Concentrations were generally at trace levels, with chloroform detected more frequently and at higher concentrations in groundwater. Chloroform has been detected in surface waters at levels as high as 85 ppb (USGS 2003). In another survey, chloroform was detected in surface waters of three rivers in Arizona at 5.4, 3.8, and 2.3 ppb (Rostad et al. 2000).

Table 5-8. Summary of Concentrations of Chloroform (ppb) Measured in Surface and Groundwater Across the United States

Year	Average	Maximum	Number of samples	Percent detected
Surface water				
2018	0.68	4.5	510	2.2
2019	0.29	1.6	459	3.1
2020	0.58	2.6	198	5.1
2021	0.71	5.7	499	3.0
2022	1.24	4.3	351	2.6
2023	0.15	0.49	71	14
Groundwater				
2018	4.2	120	1,323	21
2019	1.4	29	2,022	21
2020	1.5	25	1,689	34
2021	1.6	43.6	2,142	36
2022	1.4	38	3,020	22
2023	2.4	324	1,970	20

Source: WQP 2024

In a survey of principal aquifers in the United States between 1991 and 2010, the U.S. Geological Survey (USGS 2015a, 2015b) reported the number of aquifers in which volatile chemicals, including chloroform, were detected at a preselected benchmark concentration of >0.2 ppb. Multiple samples were taken per aquifer with the intention to determine the frequency and variation of chloroform detection in each aquifer. During this period, chloroform was detected at >0.2 ppb in 10 of the 17 shallow aquifers beneath agricultural land, and in 19 of the 22 aquifers beneath urban land (USGS 2015a, 2015b). The number of samples per site with chloroform present at concentrations above the benchmark of 0.2 ppb ranged from 0.75 to 8.11% for agricultural aquifers and from 2.27 to 55.00% beneath urban land. In another study, chloroform was detected in 48.1% of aquifers sampled below urban areas; 13.9% had concentrations above the benchmark of 0.2 ppb (Squillace et al. 2004). For urban and untreated rural wells, samples with concentrations >0.2 ppb were 26.4 and 7.3%, respectively (Squillace et al. 1999, 2004). Detection frequency was associated with redox conditions of the groundwater, with greater chloroform concentrations associated with increased levels of oxygen in the water. This is due to increased stability of chloroform in water with larger concentrations of oxygen, compared to low-oxygen conditions.

Between 1991 and 2010, chloroform was detected above the benchmark of 0.2 ppb in 28 of the 40 areas of principal aquifers in the United States used for drinking water (USGS 2015a, 2015b). The number of samples with concentrations above 0.2 ppb within these areas were relatively low, ranging from 0.73 to 23.68%. Between 1986 and 2001, chloroform was detected at >0.2 ppb in 11.4% of 1,092 public wells and 5.2% of 2,400 domestic (private) wells used for drinking water across the United States (USGS 2006). Measured concentrations ranged from approximately 0.008 to 23 ppb chloroform in public wells and from approximately 0.002 to 75 ppb chloroform in domestic wells. An independent analysis of these data reported that detections in domestic wells were associated with dissolved oxygen content, with higher probability of detecting chloroform associated with higher dissolved oxygen (Rowe et al. 2007). Consistent with USGS data, another more recent large-scale study by Bexfield et al. (2022a) detected chloroform at >0.2 ppb in 93 of 1,537 (6.1%) of samples collected between 2013 and 2019 from 1,531 wells and 6 springs used for drinking water in the United States (calculated from supplemental data presented in Bexfield et al. (2022b)). In total, chloroform was only detected above the limit of detection (0.015 ppb) in 25% of wells, with concentrations ranging from <0.03 to 46.82 ppb (Bexfield et al. 2022b).

Other U.S. studies have also detected chloroform in drinking water. In Florida, 7.1 ppb chloroform was detected in tap water in Dade County and 14.8 ppb was detected in tap water from Broward County (Gibbons and Laha 1999). Households in the Lower Rio Grande valley had median levels of 5.0 and 4.1 ppb chloroform detected in spring and summer of 1993, respectively (Berry et al. 1997). Concentration ranges for these two time periods were 1.1–26.1 and 2.0–18.2 ppb, respectively. Of the 70 residential wells sampled in the Piedmont and Upper Coastal Plain regions of South Carolina, only three wells had detectable chloroform: at 0.9, 1.3, and 7.5 ppb (Aelion and Conte 2004). In a statewide study of Arizona residential drinking water, chloroform was detected at a mean of 2.60 ppb in tap water and 1.30 ppb in nontap water (Sofuoglu et al. 2003). For populations on the Arizona-Mexican border, 0.39 ppb mean chloroform was detected in tap water and 0.74 ppb was detected in non-tap water (Sofuoglu et al. 2003).

Limited recent data are available for chloroform in water near hazardous waste sites. Chloroform was not detected in surface or groundwater collected during a 2013 sampling campaign at Palermo Wellfield Superfund Site (WQP 2024). Chloroform was also not detected in groundwater during a 2011 sampling campaign of the Boomsnub Superfund Site (WQP 2024). Historical levels of chloroform in drinking water derived from wells near hazardous waste dumps in the 1980s ranged from 0.3 to 1,890 ppb (Clark et al. 1982; Dewalle and Chian 1981).

A review of ATSDR public health assessments completed between 1994 and 2009 identified 33 assessments with chloroform detected on site (Burk and Zarus 2013). Chloroform was detected in groundwater at 15 sites ranging from 0.3 to 134 μg/L.

In addition to drinking water, chlorinated oxidants reacting with organic materials will lead to the formation of chloroform in swimming pools. Reported concentrations in samples from public pools fall in a range of 32–207 ppb (Kanan et al. 2015). In poorly tended or very crowded pools, where there are large inputs of organic materials or heavy use of chlorinating agents, chloroform formation increases (Kanan et al. 2015). Chloroform production in swimming pools can be increased where the pools are treated with copper-containing algicides. In tests on chlorinated water using various doses of chlorine, cupric salts (with various anions), and varying levels of humic acid (Barnes et al. 1989), chloroform concentrations after a given reaction time were generally ≥50% higher in samples treated with copper, which acts as a catalyst in the reactions with the humic acids.

Chloroform at 0.25 ppb has been found in rainwater collected in Los Angeles, California, in 1982 (Kawamura and Kaplan 1983). Chloroform has been detected in seawater between 0.0016 and 1.090 ppb (Ohligschläger et al. 2019).

5.5.3 Sediment and Soil

Very limited recent soil and sediment analysis for chloroform has been conducted. Chloroform was detected in 4 of 365 sediment samples collected from the Great Lakes in 2020–2021; the maximum concentration was 72 ppb (WQP 2024). However, no chloroform was detected in 28 sediment samples collected in Texas in 2019. Historically, chloroform has been found in sediment samples from the three passes of Lake Pontchartrain, Louisiana, at concentrations ranging from 0.0017 to 0.018 ppb (wet weight basis) (Ferrario et al. 1985).

Chloroform was found at concentrations ranging from 0.030 to 0.080 ppb (dry weight basis) in sediment samples exposed to chlorinated electrical power plant cooling water; the control samples that were not exposed to cooling water contained nearly the same amounts of chloroform (Bean et al. 1985). Chloroform was detected in sediments collected in 2010 at the BP Deep Water Horizon oil spill, between 5.33 and 44.3 ppb (WQP 2024). Chloroform was not detected in sediment samples collected in 2021 at the Palermo Wellfield Superfund Site (WQP 2024).

Soil data are primarily limited to industrial sites. The only ambient soil sampling data available were seven samples collected in Montana in 2023; chloroform was not detected (detection limits are 0.087–0.13 mg/kg). Chloroform was detected in four soil samples collected in 2011 from the Salt Chuck Mine in Alaska at 12–48 ppb (WQP 2024). Chloroform was not detected in soil samples collected in 2015 from the Gay Mine Superfund Site (WQP 2024). It can be predicted that chloroform contamination occurs at hazardous waste sites where chloroform-containing leachate moves through the soil to groundwater. An explanation of the lack of data results from the fact that any chloroform in the soil is expected to either rapidly volatilize or leach. Laboratory studies using a variety of different soil types document the effectiveness of volatilization in removing chloroform from soils (Park et al. 1988).

5.5.4 Other Media

Chloroform has been detected in various foods and beverages at trace concentrations (in the ppb range). A summary of the available data is provided in Table 5-9. The most recent data are from a Canadian market study conducted in 2015 (Cao et al. 2024). Chloroform was detected in 37 of the 159 food and beverage samples, with butter, tap water, baking powder, cream, and cheese having the highest chloroform concentrations. In an older American study, dairy products also had the highest chloroform concentrations (Fleming-Jones and Smith 2003). This was likely due to the use of cleaning and disinfecting solvents containing chlorine, which form chloroform when residual cleaning products used on the processing equipment come in contact with the organic material in the dairy products. Rinsing equipment pre- and post-cleaning reduces this chloroform contamination source (Fleming-Jones and Smith 2003).

Fleming-Jones and Smith (2003) reported chloroform in foods such as produce and meat that were >100 ppb in total VOCs. Pork bacon, beef frankfurters, fast food chicken nuggets, and avocado were some of the foods that met the total VOC criteria and contained chloroform. Additionally, chloroform was detected in butter and cheese samples, which is in accord with reporting by Cao et al. (2024). Both studies reported chloroform levels (3–14 ppb) in beef frankfurters or wieners and sausages (Cao et al. 2024, Fleming-Jones and Smith 2003). There are several reasons why the studies may differ in the samples containing chloroform, especially as Fleming-Jones and Smith (2003) only reported chloroform levels in foods that met the VOC criteria. In addition, the studies prepared food samples differently, with Fleming-Jones and Smith (2003) cooking foods that are normally cooked and Cao et al. (2024) using a mixture of raw and cooked foods in their samples. It is possible that different food preparation methods may have contributed to the difference in results.

Residual chlorine in potable water can react with organic material to form chloroform, and chloroform has been observed to contaminate food and beverages during preparation (e.g., brewing tea or coffee, cooking soups, boiling vegetables or meat) and rinsing with chlorinated water (Huang and Batterman 2009, 2010). Chloroform may be introduced into commercial foodstuff and beverages via use of chlorinated water during food or beverage processing (Huang and Batterman 2009).

Table 5-9. Summary of Chloroform Measured in Food and Beverages in the United States (ppb)

Product	Average	Range	Source
Milk, whole	5.6		Cao et al. 2024[a]
Milk, 2%	1.9		
Milk, 1%	1.3		
Milk, skim	0.5		
Cream	26		
Ice cream	10		
Yogurt	3.5		
Cheese	21		
Cheese, cottage	13		
Cheese, processed	12		
Butter	58		
Chocolate milk, 1%	2.3		
Butter milk, 1%	0.6		
Soups, creamed, canned	4.1		
Apple juice, canned	3.6		
Citrus juice, frozen	0.6		
Citrus juice, canned	7.6		
Grape juice, bottled	2.4		
Fruit drinks (cocktails)	4.1		
Alcoholic drinks, beer	0.24		
Soft drinks, canned	2.9		
Soy beverage, fortified	1.6		
Tap water, kitchen	29		
Tap water, sample area	23		
Salad dressing	3.5		
Baby formula, milk base	2.8		
Syrup	2.2		
Nuts	3.9		
Condiments	0.6		
Salt	4		
Baking powder	26		

Table 5-9. Summary of Chloroform Measured in Food and Beverages in the United States (ppb)

Product	Average	Range	Source
Vanilla extract	1.7		
Soy sauce	0.8		
American cheese	–	11–54	Fleming-Jones and Smith 2003[b]
Cheddar cheese	–	3–107	
Cream cheese	–	38–100	
Margarine	–	7–14	
Butter	–	35–83	
Sour cream	–	14–176	
Cheese pizza	–	3–11	
Cheese and pepperoni pizza	–	2–6	
Ground beef	–	2–6	
Pork bacon	–	2–12	
Beef frankfurters	–	3–14	
Tuna, canned in oil	–	4	
Eggs, scrambled	–	5–13	
Quarter pound hamburger, cooked	–	2–14	
Chicken nuggets, fast food	–	2–16	
Cheeseburger, quarter pound	–	2–15	
Bologna	–	5–15	
Banana	–	8	
Peanut butter	–	2–8	
Raw avocado	–	3–15	
Popcorn, popped in oil	–	2–15	
Blueberry muffin	–	8–15	
Raw orange	–	4–6	
Potato chips	–	3–12	
Apple pie, fresh/frozen	–	9–19	
French fries, fast food	–	2–3	
Olive/safflower oil	–	4	
Mixed nuts	–	4–5	
Chocolate cake with icing	–	3–16	
Fruit-flavored cereal	–	2	
Cola, carbonated beverage	–	11–27	
Sweet roll/danish	–	2–12	
Fruit-flavored sherbet	–	0–27	
Popsicle	–	6–18	
Sandwich cookies	–	2–14	

Table 5-9. Summary of Chloroform Measured in Food and Beverages in the United States (ppb)

Product	Average	Range	Source
Chocolate chip cookies	–	3–4	
Graham crackers	–	5–12	
Sugar cookies	–	3–10	
Cake doughnuts with icing	–	2–6	
Natural spring bottled water (two brands sampled)	4.0; 3.8	–	Gibbons and Laha 1999

[a] Only positive detections were reported.

[b] Only foods with at least 100 ppb of any of the studied volatile organic compounds (VOCs) were reported.

Chloroform was not reported by the FDA's Total Diet Study in recent years. Previously, 41% of 231 samples of various foods contained chloroform at levels ranging from 4 to 312 ppb and 55% of 549 samples contained between 2 and 830 ppb chloroform (Daft 1988, 1989).

Chloroform was detected in volatile emissions at trace levels in baby wipes (56% of samples), baby diapers (67% samples), and adult diapers (50% of samples) (Lin et al. 2023). The maximum detection was 12 ng/g in a baby diaper. Chloroform may be present as an inadvertent contaminant or byproduct from manufacturing (Lin et al. 2023).

Since chloroform is highly volatile and shows little tendency to bioconcentrate or bioaccumulate in higher life forms such as fish, it is not ordinarily included in the types of persistent pollutants that are the focus of state fish consumption advisory programs. In a limited sampling survey conducted by the city and county of Honolulu between 2010 and 2014, chloroform was not detected in three species of marine fish: *Myripristis berndti*, *Selar crumenophthalmus*, and *Lutjanus kasmira* (WQP 2024).

5.6 GENERAL POPULATION EXPOSURE

The general population is likely to be exposed to chloroform through drinking water and beverages, eating food, inhaling contaminated air, and through dermal contact with water containing chloroform as a disinfection byproduct (e.g., while showering, bathing, cleaning, washing, swimming). All humans are expected to be exposed to at least low levels of chloroform. The most common chloroform exposures relate to chloroform generated when organic materials interact with chlorinated oxidants (e.g., chlorine or hypochlorous acid) widely used to purify water or remove pathogens from waste materials. Exposure to commercially produced chloroform is expected to be less common.

Accurate, current estimates of the daily intake of chloroform by various exposure routes are not available, or possible, due to the lack of appropriate monitoring data. Typical levels of atmospheric chloroform in remote, ambient, and source-dominated areas are 0.008–0.0098, 0.018–0.04, and 0.51–0.81 ppbv, respectively (EPA 1982, 1988b; McCarthy et al. 2006). Exposure via ingestion of contaminated drinking water is expected to be extensive since most U.S. community drinking-water supplies are chlorinated. Typical levels in drinking water range from 0.002 to 75 ppb (Aelion and Conte 2004; Berry et al. 1997; Gibbons and Laha 1999; Sofuoglu et al. 2003; USGS 2006). Levels in food have been well characterized, although data are not recent; the foods with the highest levels of chloroform are typically dairy products (Cao et al. 2024; Fleming-Jones and Smith 2003). Chloroform contamination is believed to be from water used for cooking and rinsing during food preparation, such as beverages and foods prepared by boiling water, including tea (3–67 ppb), coffee (3–13 ppb), rice (9 ppb), and soup (0.4–3 ppb) (Huang and Batterman 2009, 2010).

Generally low personal exposures are expected when not near a chloroform source. Personal air monitoring studies of married women in Pennsylvania and New Jersey were conducted for those who worked in smoking or nonsmoking environments and were married to a smoker or nonsmoker (Heavner et al. 1996). Mean personal air concentrations were comparable in nonsmoking homes (0.60 $\mu g/m^3$) and smoking homes (0.85 $\mu g/m^3$). Similarly, mean personal air concentrations in workplaces were comparable in nonsmoking (0.88 $\mu g/m^3$) and smoking environments (0.91 $\mu g/m^3$). Based on these measurements, daily median exposures to chloroform were calculated to be between 0.06 and 0.30 $\mu g/m^3$ for the different combinations of smoking and nonsmoking work and home environments (Heavner et al. 1996). In another personal air monitoring study of graduate students in a mixed-use university art building (mainly in areas where silk screen printing and cleaning occurred), median concentrations over 3-hour sampling periods were below the detection limits, which were reportedly between 0.5 and 1.5 ppb (Ryan et al. 2002).

Chloroform can be expected to exist in virtually all chlorinated drinking-water supplies. The main source of chloroform found in municipal drinking water is the chlorination of naturally occurring humic materials found in raw-water supplies (Ohligschläger et al. 2019). Factors that can increase the amount of chloroform in drinking water include seasonal effects (high summer values) and increased contact time between chlorine and humic material. Sources of water with high humic material content will contain higher levels of chloroform. The chloroform concentration in drinking water plants increases as the treated water moves through the distribution system, which contains humic or other organic material that reacts with the chlorine used for disinfection (Kasso and Wells 1981). Drinking water derived from groundwater may contain higher levels of chloroform than normally encountered in drinking water derived from surface water. Microplastics in water may leach DOM, which promotes chloroform formation during water treatment (Chen et al. 2024; Yan et al. 2024). Chloroform ingestion from drinking water may be decreased in homes which utilize household reverse osmosis filtering systems or pitchers with activated carbon and ion exchange resin filters (Carrasco-Turigas et al. 2013). Reductions ranged from 56 to 91%, with greater removal observed for the pitcher filters.

Chloroform exposure in air may occur from chlorinated water, as chloroform is readily volatilized. Upper bound normalized daily inhalation from bathroom air during a 12-minute shower scenario was calculated to be 0.63–0.50 μg chloroform/day/μg chloroform in a liter of water (Kerger et al. 2005). Inhalation of chloroform during the 20-minute period immediately after the shower was calculated to be 0.56–0.58 μg chloroform/day/μg chloroform in a liter of water. Likewise, chloroform may volatilize during other domestic activities which utilize chlorinated water. Approximately two-thirds of chloroform formed during simulated normal dishwasher usage were released to the air (Olson and Corsi 2004). During the last 40 minutes of the 50-minute dishwasher cycle, chloroform was detected in the headspace air of all tested dishwashers at concentrations ranging from 0.01 to 36.1 μg/L. In the first 10 minutes, concentrations ranged from 0.04 to 6.3 μg/L in 8/10 dishwashers; chloroform levels were below the limit of detection (0.008 μg/L) for the other 2 dishwashers.

ATSDR's three-compartment Shower and Household-Use Exposure (SHOWER) model predicts air concentrations in the shower stall, bathroom, and main house throughout the day by estimating the contribution from showering or bathing and the contribution from other water sources in the house, such as the dishwasher, clothes washer, and faucets. This information, along with human activity patterns, is used to calculate a daily time-weighted average exposure concentration via inhalation exposure and from dermal uptake from skin contact. ATSDR's SHOWER model is available by sending a request to showermodel@cdc.gov. Using median treated water levels as discussed in Section 5.5.2 and representative outdoor air levels discussed in Section 5.5.1, reasonable maximum exposure (RME) levels for chloroform were calculated for different exposure groups and are presented in Table 5-10.

Table 5-10. Reasonable Maximum Exposure Daily Inhalation Dose in µg/kg/day and Administered Dermal Dose of Chloroform for the Target Person

Exposure group	Inhalation (µg/m^3)	Dermal (µg/kg/day)
Birth–<1 year	2.5	0.013
1–<2 years	2.5	0.012
2–<6 years	2.5	0.010
6–<11 years	2.5	0.0083
11–<16 years	2.5	0.0067
16–<21 years	2.5	0.0062
Adult	2.5	0.0061
Pregnant and breastfeeding women	2.5	0.0061

Source: ATSDR 2022b

An exposure study of 50 mothers was conducted in two areas with relatively high chloroform concentrations in municipal water compared to national averages, Corpus Christi, Texas, and Cobb County, Georgia, to determine effects to blood chloroform concentrations after showering. Median pre-showering levels were 25 and 70 pg/mL, respectively, and post-showering levels were significantly increased at 57 and 280 pg/mL, respectively (Lynberg et al. 2001). Monitoring of return to baseline was not included as part of the study.

Individuals may also be exposed to chloroform while swimming or lounging in chlorinated pools or spas through inhalation of volatilized chloroform in the air and dermal contact with the water. Monitoring studies have reported mean air concentrations of chloroform ranging from 2.66 to 105.73 ppbv at indoor recreational swimming pool facilities (Font-Ribera et al. 2010; Kanan et al. 2015; Nitter and Svendsen 2019; Sa et al. 2011). Pool water chloroform levels ranged from 3.04 to 114.5 µg/L (Aggazzotti et al. 1995; Cammann and Hübner 1995; Font-Ribera et al. 2010).

The Centers for Disease Control and Prevention (CDC) creates ongoing assessments on human exposure to environmental chemicals derived from data obtained from NHANES. The biomonitoring data reports levels of chloroform from a sample of people who represent the noninstitutionalized, civilian U.S. population during 2-year study periods conducted between 2011 and 2018. The data are presented in Table 5-11. Detectable levels of chloroform in blood are expected to reflect recent exposure; blood levels of chloroform can raise 2–4 times over baseline levels and return to normal rapidly after 1–2 hours (CDC 2022a, 2022b).

Table 5-11. Geometric Mean and Selected Percentiles of Chloroform Blood Concentrations (pg/mL) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) (2011–2018)

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample Size
			50th	75th	90th	95th	
Total	2011–2012	6.32 (5.24–7.62)	6.06 (4.73–8.00)	12.6 (10.1–16.2)	22.5 (19.8–27.0)	33.5 (27.7–40.2)	2,589
	2013–2014	*	8.00 (<LOD–10.0)	16.0 (14.0–19.0)	27.0 (24.0–32.0)	40.0 (33.0–50.0)	3,136
	2015–2016	*	9.00 (8.00–11.0)	17.0 (15.0–20.0)	30.0 (28.0–33.0)	47.0 (42.0–54.0)	2,989
	2017-2018	*	8.00 (<LOD–9.00)	14.0 (11.0–17.0)	24.0 (20.0–30.0)	34.0 (30.0–42.0)	2,858
Age 12–19 years	2011–2012	5.08 (4.10–6.29)	4.76 (3.42–6.21)	9.22 (7.76–12.4)	20.8 (15.5–25.8)	28.7 (21.5–34.6)	487
	2013–2014	*	<LOD	14.0 (12.0–18.0)	28.0 (21.0–33.0)	35.0 (28.0–53.0)	588
	2015–2016	*	<LOD	14.0 (11.0–17.0)	26.0 (19.0–35.0)	37.0 (26.0–53.0)	537
	2017-2018	*	<LOD	11.0 (9.00–15.0)	23.0 (18.0–30.0)	32.0 (26.0–44.0)	471
Age 20+ years	2011–2012	6.54 (5.41–7.89)	6.26 (4.91–8.40)	12.9 (10.5–16.7)	23.2 (19.9–27.3)	34.1 (28.0–43.1)	2,102
	2013–2014	*	8.00 (<LOD–10.0)	16.0 (14.0–19.0)	27.0 (25.0–32.0)	41.0 (34.0–50.0)	2,548
	2015–2016	*	9.00 (8.00–11.0)	17.0 (16.0–20.0)	30.0 (28.0–35.0)	47.0 (40.0–57.0)	2,452
	2017-2018	*	8.00 (<LOD–9.00)	14.0 (11.0–17.0)	24.0 (20.0–30.0)	34.0 (30.0–43.0)	2,387
Males	2011–2012	6.27 (5.14–7.64)	6.18 (4.74–7.98)	12.3 (10.1–15.1)	22.3 (18.5–28.1)	35.6 (27.3–47.2)	1,307
	2013–2014	*	<LOD	16.0 (14.0–19.0)	26.0 (22.0–35.0)	38.0 (31.0–51.0)	1,512
	2015–2016	*	9.00 (<LOD–11.0)	17.0 (16.0–19.0)	31.0 (27.0–37.0)	51.0 (43.0–58.0)	1,485
	2017-2018	*	8.00 (<LOD–9.00)	14.0 (11.0–17.0)	26.0 (20.0–32.0)	35.0 (30.0–44.0)	1,390
Females	2011–2012	6.38 (5.25–7.75)	5.85 (4.62–8.16)	12.9 (9.94–17.9)	22.9 (19.4–28.0)	32.1 (26.6–41.0)	1,282
	2013–2014	*	9.00 (<LOD–10.0)	16.0 (14.0–19.0)	28.0 (25.0–34.0)	41.0 (33.0–52.0)	1,624
	2015–2016	*	9.00 (8.00–10.0)	17.0 (15.0–20.0)	29.0 (26.0–33.0)	42.0 (35.0–55.0)	1,504
	2017-2018	*	8.00 (<LOD–9.00)	14.0 (11.0–17.0)	23.0 (20.0–29.0)	32.0 (29.0–39.0)	1,468
Mexican Americans	2011–2012	6.14 (4.94–7.62)	5.91 (4.76–7.80)	11.7 (9.94–15.0)	23.1 (15.0–33.1)	32.5 (24.9–46.4)	269
	2013–2014	*	<LOD	9.00 (<LOD–14.0)	16.0 (12.0–22.0)	22.0 (16.0–33.0)	503
	2015–2016	*	8.00 (<LOD–10.0)	16.0 (12.0–21.0)	28.0 (23.0–35.0)	41.0 (32.0–46.0)	542
	2017-2018	*	<LOD	10.0 (8.00–16.0)	19.0 (14.0–27.0)	27.0 (19.0–36.0)	418
Non-Hispanic blacks	2011–2012	8.64 (6.08–12.3)	8.28 (5.22–13.6)	17.0 (10.8–27.0)	30.8 (20.0–45.2)	42.6 (32.3–56.5)	697
	2013–2014	*	10.0 (<LOD–14.0)	18.0 (14.0–24.0)	30.0 (23.0–40.0)	40.0 (30.0–65.0)	595
	2015–2016	11.6 (10.2–13.2)	10.0 (9.00–13.0)	18.0 (15.0–21.0)	28.0 (23.0–34.0)	38.0 (33.0–46.0)	619
	2017-2018	10.8 (9.40–12.5)	9.00 (<LOD–11.0)	16.0 (13.0–20.0)	27.0 (24.0–32.0)	39.0 (27.0–64.0)	635

Table 5-11. Geometric Mean and Selected Percentiles of Chloroform Blood Concentrations (pg/mL) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) (2011–2018)

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample Size
			50th	75th	90th	95th	
Non-Hispanic whites	2011–2012	5.58 (4.50–6.90)	5.35 (3.85–7.41)	11.2 (7.93–14.6)	21.1 (16.7–25.4)	28.0 (24.0–37.0)	901
	2013–2014	*	9.00 (<LOD–11.0)	17.0 (14.0–20.0)	28.0 (23.0–36.0)	41.0 (32.0–57.0)	1,266
	2015–2016	*	9.00 (<LOD–11.0)	17.0 (15.0–19.0)	30.0 (27.0–33.0)	49.0 (37.0–58.0)	989
	2017-2018	*	<LOD	14.0 (11.0–16.0)	24.0 (20.0–29.0)	34.0 (30.0–38.0)	976
All Hispanics	2011–2012	7.85 (6.08–10.1)	8.03 (5.44–10.9)	15.3 (10.6–22.3)	26.3 (21.2–35.6)	37.2 (31.6–47.0)	546
	2013–2014	*	<LOD	12.0 (<LOD–17.0)	23.0 (17.0–29.0)	30.0 (23.0–45.0)	806
	2015–2016	*	9.00 (<LOD–11.0)	18.0 (14.0–22.0)	32.0 (27.0–36.0)	46.0 (36.0–52.0)	909
	2017-2018	*	<LOD	12.0 (9.00–17.0)	22.0 (16.0–30.0)	31.0 (21.0–42.0)	678
Asians	2011–2012	7.71 (6.25–9.51)	7.99 (6.25–10.0)	14.2 (11.1–18.1)	22.9 (18.8–30.2)	31.3 (24.2–43.1)	371
	2013–2014	*	<LOD	14.0 (10.0–20.0)	25.0 (17.0–39.0)	39.0 (27.0–47.0)	361
	2015–2016	*	9.00 (<LOD–11.0)	16.0 (11.0–23.0)	32.0 (22.0–46.0)	47.0 (34.0–62.0)	346
	2017-2018	*	8.00 (<LOD–18.0)	18.0 (9.00–29.0)	29.0 (18.0–52.0)	41.0 (26.0–71.0)	405

[a] LODs for survey years 2011–2012, 2013–2014, 2015–2016, and 2017–2018 are 2.1, 8, 8, and 8 pg/mL, respectively.

* = not calculated; proportion of results below limit of detection was too high to provide a valid result; CI = confidence interval; LOD = limit of detection

Source: CDC 2022a, 2022b

Limited other biomonitoring data are available. One study identified the VOCs present in amniotic fluid of 76 pregnant French women during the second trimester of their healthy pregnancy (Minet-Quinard et al. 2023). Chloroform was detected with a 44.7% frequency in the women's amniotic fluid.

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Limited current data were located regarding occupational exposure to chloroform. Although some of the exposure levels encountered in workplaces may be comparable to exposure that workers receive in their own homes, there are probably many specific jobs that expose workers to significantly higher levels of chloroform. These occupations likely include work at or near source-dominated areas such as chemical plants and other facilities that manufacture or use chloroform, operation of chlorination processes in drinking-water plants, work at or near wastewater-treatment plants and paper and pulp plants, and other facilities where large amounts of chloroform are released (e.g., hazardous and municipal-waste incinerators). Additionally, workers who also live in communities near production/use facilities (e.g., fence line communities) have an even greater potential for exposure. A maximum level of 3.8 ppbv was found in the air at an activated sludge wastewater treatment plant (Lurker et al. 1983). Maintenance workers, attendants, and lifeguards at indoor pools and spas may encounter concentrations between 32 and 207 ppb in water or 2.66–105.73 ppbv in air (Ahmadpour et al. 2022; Kanan et al. 2015; Nitter and Svendsen 2019; Sekar et al. 2022). Air may be a significant route of exposure for these workers, as one study found no significant difference between chloroform concentrations collected in close proximity to the pool surface (0.5 m) compared to approximate worker height (1.5 m) (Ahamdpour et al. 2022). For sewage/effluent treatment operators working in pulp, paper, and paper product mills, exposures to 15–1,670 ppb chloroform in water have been reported in the United States (Teschke et al. 1999). While occupational exposures are expected to be predominantly inhalation, there is potential for exposure via dermal contact with vapor or liquid.

Persons who use tap water often, especially if it is heated and/or sprayed (e.g., water used for cleaning, washing clothes and dishes, showering, and cooking), may also be exposed to higher than background levels. Levels in personal air samples as high as 22 and 11 ppb have been measured during household cleaning activities and showering (Wallace et al. 1987b). Persons using certain cleaning agents and pesticides in enclosed spaces with poor ventilation or persons working where these materials are used may be exposed to relatively high levels of chloroform. While the use of activated carbon filters may provide some reduction in the tap water levels for cold water, such filters are not effective with hot water where the elevated temperatures will induce volatilization from the filter media.

Individuals employed as cleaners (e.g., janitors, hotel housekeeping, domestic staff) form an occupational group that may have increased risk of exposure to chloroform (Lin et al. 2022; Wolfe et al. 2020). The potential for chloroform exposure in these occupations is elevated not only due to increased usage of disinfected water but also from chloroform generated during use of chlorine-containing disinfectants, such as bleach (Bruchard et al. 2023; Odabasi 2008; Lin et al. 2022). A particularly vulnerable group may be Hispanic women, who make up 58.9% of domestic house cleaners in the United States (Wolfe et al. 2020). Switching from traditional cleaning products to products labelled as "green" cleaning products has been shown to reduce chloroform exposure during domestic cleaning (Harley et al. 2021). Geometric mean personal air concentrations of chloroform while cleaning were 0.5 and 0.066 ppb while using traditional and "green" cleaning products, respectively.

Increased non-occupational exposure to chloroform is likely to occur from increased contact with chlorinated water. People who regularly swim in chlorinated pools, such as competitive swimmers, may have increased exposure to chloroform through inhalation and dermal contact with the water. Exposure levels are higher in heated, indoor pools, compared to outdoor training facilities (Aiking et al. 1994). In addition to increased frequency of exposure, kinetic studies show that increased physical exercise while in a chlorinated pool increases absorption of chloroform during the exposure period (Cammann and Hübner 1995). Several factors likely impact the chloroform levels reported in air from indoor swimming facilities, which ranged from 3.99 to 205.6 ppbv (Ahmadpour et al. 2022; Font-Ribera et al. 2010; Nitter and Svendsen 2019; Sa et al. 2011). These may include pool temperature, concentration of chlorine used, how crowded the pool is, and ventilation systems used.

Persons living in certain source-dominated areas may be at risk for higher than background exposures to chloroform. These may include persons living near industries and facilities that manufacture and use chloroform, municipal and industrial wastewater-treatment plants and incinerators, landfills and other waste sites, and paper and pulp plants. Likewise, persons who derive their drinking water from groundwater sources contaminated with chloroform from hazardous waste sites may have higher exposures. Additionally, exposure to chloroform in air may be increased in homes with vapor intrusion problems, and sewer lines below residences should be investigated as a potential source of chloroform emissions (ATSDR 2016; McHugh et al. 2017).

Individuals with inhalant substance use disorder that repeatedly and intentionally self-administer inhalants to achieve intoxication may have increased risk of exposure to chloroform depending upon the products abused. Chloroform is among the many chemicals in commonly abused products (Howard et al. 2011).