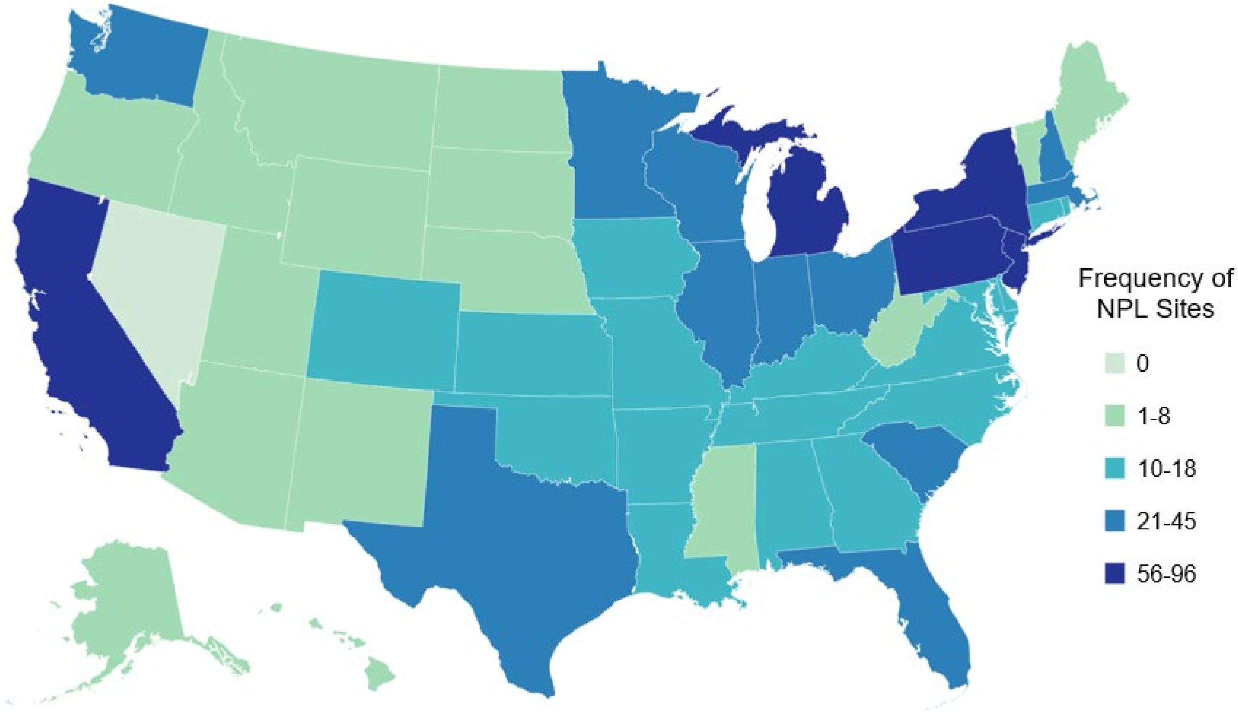
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Benzene has been identified in at least 982 of the 1,868 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2022a). However, the number of sites in which benzene has been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Of these sites, 977 are located within the United States, 2 are located in the Virgin Islands, and 3 are located in Puerto Rico (not shown).

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



Source: ATSDR 2022a

- The general population is most likely to be exposed to trace levels of benzene in ambient air and, typically at higher concentrations, in indoor air. Benzene is ubiquitous in the atmosphere from anthropogenic and natural sources, but concentrations have decreased in the past decades. Indoor air sources include evaporative emissions from cars in attached garages and cooking on gas stoves. Benzene has also been detected in surface and groundwater.

- Activities such as pumping gasoline and smoking increase inhalation benzene exposure.

- Benzene has been detected in the parts per billion range in some foods from cooking processes or formation from added preservatives, and rarely (<1%) in municipal water. These are not expected to be major exposure pathways.

- Environmental exposure from air, surface water, and drinking water may be increased for people living near hazardous waste sites.

- Benzene readily volatilizes to air from water and is mobile in soils and will migrate to groundwater. Bioaccumulation in biota is not expected to occur to a significant degree.

- Residence times in the atmosphere ranged from hours to days based on indirect photolysis. Indirect photolysis may also be a transformation mechanism in surface water. Benzene is readily biodegraded in aerobic conditions (e.g., surface water and soil) and is not readily degraded under anerobic conditions (e.g., groundwater and subsurface sediments).

Benzene is released to the environment by both natural and industrial sources, although the anthropogenic emissions are undoubtedly the most important. Emissions of benzene to the atmosphere result from gasoline vapors, auto exhaust, and chemical production and user facilities. EPA's estimate of nationwide benzene atmospheric emissions from various point and non-point sources was approximately 152,000 metric tons in 2020 (EPA 2020a). Benzene is released to water and soil from industrial discharges, landfill leachate, and gasoline leaks from underground storage tanks.

Chemical degradation reactions, primarily reaction with hydroxyl radicals, limit the atmospheric residence time of benzene to only a few days. Under certain conditions, atmospheric residence time may be as brief as a few hours. Benzene released to soil or waterways is subject to volatilization, photooxidation, and biodegradation. Biodegradation, principally under aerobic conditions, is an important environmental fate process for water- and soil-associated benzene.

Benzene is ubiquitous in the atmosphere. It has been identified in air samples of both rural and urban environments and in indoor air. Although a large volume of benzene is released to the environment, environmental levels are low due to degradation processes. Benzene partitions mainly into air (99.9%) and inhalation is the dominant pathway of human exposure accounting for >99% of the total daily intake of benzene (Hattemer-Frey et al. 1990; MacLeod and MacKay 1999).

The general population is exposed to benzene primarily by tobacco smoke (both active and passive smoking) and by inhaling contaminated air, particularly in areas with heavy motor vehicle traffic and around filling stations, and in some cases, poorly ventilated indoor air. Indoor air benzene pollution sources include gas stoves and ovens, evaporative emissions from cars in attached garages, and fuel or wood-based heat sources (e.g., fireplaces). Use of contaminated tap water for cooking, showering, etc., can also be a source of inhalation exposure since benzene can volatilize from water. Air around manufacturing plants that produce or use benzene and air around landfills and hazardous waste sites that contain benzene are additional sources of exposure.

Another source is from smoking, which was found to be the largest anthropogenic source of direct human exposure to benzene (Duarte-Davidson et al. 2001; Hattemer-Frey et al. 1990). In the United States, tobacco cigarette smoking was found to be the predominant source of increased blood benzene concentrations in which people who smoked at least one cigarette per day had a mean level of 0.140 µg/L benzene compared to people who smoked less than one cigarette per day, which includes nonsmokers, and had a mean level below detection (<0.024 µg/L) (Chambers et al. 2011).

Exposure to benzene can also result from ingestion of contaminated food or water but this is not expected to be a major exposure pathway. Compared to inhalation, dermal exposure accounts for a minor portion of the total exposure of the general population. Dermal exposure may occur in the general population from direct contact with gasoline (e.g., spillage while filling gas tank). Individuals occupationally exposed to benzene tend to have higher dermal doses than the general population.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Table 5-1 summarizes information on companies that reported the production, import, or use of benzene for the Toxics Release Inventory (TRI) in 2022 (TRI22 2023). 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 Benzene

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AK	18	1,000	49,999,999	1, 3, 4, 5, 7, 9, 12, 13, 14
AL	18	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
AR	8	1,000	9,999,999	1, 2, 3, 5, 6, 9, 12
AZ	17	1,000	9,999,999	1, 5, 9, 12
CA	53	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 9, 12, 13, 14
CO	26	0	99,999,999	1, 2, 3, 5, 8, 9, 12, 13, 14

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

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
CT	3	100,000	9,999,999	1, 5, 7, 9, 12
DE	1	100,000	999,999	1, 2, 3, 4, 9
FL	24	0	9,999,999	1, 2, 4, 5, 7, 9, 12, 13, 14
GA	7	10,000	9,999,999	1, 5, 8, 9, 12
GU	2	1,000	999,999	7, 9
HI	10	0	9,999,999	1, 2, 3, 4, 5, 6, 9, 12, 13, 14
IA	42	1,000	9,999,999	1, 4, 5, 7, 8, 9, 12
ID	3	1,000	999,999	1, 5, 9, 14
IL	36	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
IN	34	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
KS	22	0	499,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 14
KY	22	1,000	49,999,999	1, 3, 5, 6, 7, 8, 9, 10, 12, 13, 14
LA	79	0	499,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MA	8	1,000	9,999,999	2, 4, 7, 8, 9, 12
MD	6	1,000,000	9,999,999	9
ME	7	0	9,999,999	2, 3, 4, 7, 9, 12
MI	32	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
MN	19	1,000	49,999,999	1, 2, 3, 4, 5, 6, 7, 9, 10, 11, 12, 13, 14
MO	14	0	9,999,999	1, 5, 7, 8, 9, 10, 12
MP	2	10,000	999,999	1, 2, 3, 5, 7, 9
MS	14	10,000	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
MT	9	100,000	9,999,999	1, 2, 3, 4, 5, 6, 9, 12, 13, 14
NC	11	0	9,999,999	1, 5, 9, 12
ND	30	0	9,999,999	1, 3, 4, 5, 6, 8, 9, 11, 12, 13, 14
NE	21	1,000	9,999,999	1, 2, 3, 5, 7, 8, 9, 12, 14
NH	5	0	99	12
NJ	10	1,000	49,999,999	1, 2, 3, 4, 5, 6, 7, 9, 13, 14
NM	24	100	9,999,999	1, 3, 5, 6, 7, 8, 9, 11, 12, 13, 14
NV	3	1,000,000	9,999,999	9
NY	17	1,000	9,999,999	1, 2, 4, 5, 7, 8, 9, 12
OH	37	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
OK	32	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
OR	4	100,000	9,999,999	1, 5, 7, 9
PA	21	100	9,999,999	1, 2, 4, 5, 6, 7, 8, 9, 12, 13, 14
PR	7	100	49,999,999	2, 3, 4, 8, 9, 12, 13, 14
RI	3	100,000	9,999,999	1, 5, 7, 9, 12
SC	5	100	99,999	1, 5, 8, 12
SD	16	1,000	99,999	1, 5, 7, 9, 12, 13, 14

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

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
TN	17	0	49,999,999	1, 3, 5, 7, 8, 9, 11, 12, 13, 14
TX	264	0	10,000,000,000	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
UT	16	10,000	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
VA	6	1,000	999,999	1, 5, 9
VI	3	100,000	9,999,999	1, 5, 7, 9
WA	16	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 9, 10, 12, 13, 14
WI	19	0	9,999,999	1, 5, 7, 9, 12, 14
WV	11	1,000	9,999,999	1, 3, 5, 6, 7, 8, 9, 12, 13, 14
WY	17	100	9,999,999	1, 2, 3, 4, 6, 8, 9, 12, 13, 14

[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 2023 (Data are from 2022)

In 1825, Faraday first isolated benzene from a liquid condensed by compressing oil gas. Benzene was first synthesized by Mitscherlich in 1833 by distilling benzoic acid with lime. Benzene was first commercially recovered from light oil derived from coal tar in 1849 and from petroleum in the 1940s (Fruscella 2002). Several years after the end of World War II, the rapidly expanding chemical industry created an increased demand for benzene that the coal carbonization industry could not fulfill. To meet this demand, about 95% of commercial production of benzene shifted to the petroleum and petrochemical industries via recovery from petroleum sources (Fruscella 2002). These sources include refinery streams (catalytic reformates), pyrolysis gasoline, and toluene hydrodealkylation. Catalytic reformate is the major source of benzene, accounting for about 30% of worldwide production (Fruscella 2002).

During catalytic reforming, cycloparaffins are converted to benzene by isomerization, dehydrogenation, and dealkylation, and paraffins are converted to benzene by cyclodehydrogenation (Fruscella 2002). The type of catalyst used and process conditions determine which reaction will predominate. The benzene is recovered by solvent extraction (e.g., with tetramethylene sulfone).

Pyrolysis gasoline is a liquid byproduct produced by the steam cracking of lower paraffins (gas oil) or heavier hydrocarbons (heavy naphtha) and contains ~65% aromatics, about half of which is benzene (Fruscella 2002). Benzene is recovered from pyrolysis gasoline through hydrogenation to remove olefinic constituents, solvent extraction, and distillation for the optimization of benzene yield and the recovery of benzene.

In the toluene hydrodealkylation process, toluene or toluene/xylene mixtures are reacted with hydrogen at temperatures of 500–595°C with usual pressures of 4–6 mPa (40–60 atm), and demethylated to produce benzene and methane. Another process whereby toluene is converted to benzene and xylenes by transalkylation or disproportionation is also used for the production of benzene (Fruscella 2002). Small quantities of benzene are also produced from destructive distillation of coal used for coke manufacture. Benzene is derived from the light oil fraction produced during the coking process (Fruscella 2002).

The nationally aggregated production of benzene has held steady between $10\text{x}10^{10}$ and $20\text{x}10^{10}$ pounds between 2016 and 2019 (EPA 2022a). The companies summarized in Table 5-2 reported benzene manufacturing to the Chemical Data Reporting (CDR) Rule in 2019 (EPA 2022a). This is not an exhaustive list; companies must meet a threshold to trigger reporting to the CDR, and other manufacturers may therefore be unreported.

Table 5-2. U.S. Manufacturers of Benzene Reported to the CDR in 2019

Company	City	State[a]	Production volume (pounds)
BASF Corp.	Port Arthur	TX	NR
Chalmette Refining, LLC	Chalmette	LA	NR
Chevron	Pascagoula	MS	1,400,000,000
Citgo Holding, Inc.	Corpus Christi	TX	350,000,000
	Sulphur	LA	508,404,699
	Lemont	IL	78,000,000
Delek Us Holdings, Inc.	Big Spring	TX	NR
Deltech Corporation	Baton Rouge	LA	NR
Dynachem Inc.	Georgetown	IL	452,360
Equilon Enterprises LLC DBA Shell Oil Products US	Norco	LA	NR
Exxon Mobil Corporation	Beaumont	TX	NR
	Baton Rouge	LA	NR
	Baytown	TX	NR
Husky Energy, Inc.	Lima	OH	134,703,677
Ineos Americas, LLC	Pasadena	TX	132,800

Table 5-2. U.S. Manufacturers of Benzene Reported to the CDR in 2019

Company	City	State[a]	Production volume (pounds)
JX Nippon Chemical Texas, Inc.	Pasadena	TX	1,666,296
Koch Industries, Inc.	Corpus Christi	TX	NR
	Corpus Christi	TX	NR
Lyondell Chemical Company	Houston	TX	NR
	Channelview	TX	NR
Marathon Petroleum Company LP	Robinson	IL	251,150,313
	Texas City	TX	1,234,696,983
	Catlettsburg	KY	236,869,394
New Erie	Kalamazoo	MI	35,531
PBF Energy Inc.	Delaware City	DE	287,298,018
PBF Holding Company LLC	Oregon	OH	148,123,522
PES Ultimate Holdings, LLC	Philadelphia	PA	136,000,000
Phillips 66 Co	Belle Chasse	LA	542,863,463
	Old Ocean	TX	260,000,000
Shell Chemical LP	Norco	LA	170,437,239
	Deer Park	TX	4,330,664
SI Group USA (USAA), LLC	Bay Minette	AL	NR
The Dow Chemical Co	Plaquemine	LA	NR
Total Petrochemicals & Refining USA, Inc.	Port Arthur	TX	181,717,910
WRB Refining LP Wood River Refinery	Roxana	IL	313,717,525

[a]Post office state abbreviations used.

CDR = Chemical Data Reporting; NR = not reported

Source: EPA 2022a (data are for 2019)

5.2.2 Import/Export

Benzene is imported and exported to the United States as both the pure chemical and as a mixture of mineral fuels. Imports and exports are reported in million liters (million L) by the U.S. International Trade Commission (USITC). The import of pure benzene into the United States is dependent on domestic production and demand. Imports of benzene for consumption (from mineral fuels and pure benzene) in the United States were approximately 2,320 million L (4,480 billion pounds) in 2023, 2,054 million L (3,967 billion pounds) in 2022 1,886 million L (3,643 billion pounds) in 2021, and 2,441 million L (4,715 billion pounds) in 2020 (USITC 2024). The largest exporters of benzene to the United States in 2023 were South Korea, Canada, the Netherlands, Japan, and Brazil (USITC 2024).

As in the case of import, the export of benzene from the United States to other countries is dependent on domestic and world production and demand. Domestic exports of benzene (both pure benzene and benzene derived from mineral fuels) to other countries were approximately 125 million L (241 billion pounds) in 2023, 185 million L (357 billion pounds) in 2022, 351 million L (678 billion pounds) in 2021, and 182 million L (352 billion pounds) in 2020 (USITC 2024). These numbers are up from 23 million L (45 billion pounds) in 2001 and 4.7 million L (9.0 billion pounds) in 1993 (USITC 2024). The largest importers of benzene from the United States in 2023 were Canada, the Netherlands, Brazil, South Korea, and Mexico (USITC 2024).

5.2.3 Use

Benzene has been used extensively as a solvent in the chemical and drug industries, as a starting material and intermediate in the synthesis of numerous chemicals, and as a gasoline additive (NTP 1994).

Benzene recovered from petroleum and coal sources is used primarily as an intermediate in the manufacture of other chemicals and end products. The major uses of benzene are in the production of ethylbenzene, cumene, and cyclohexane. Ethylbenzene (52% of benzene production volume in 2008) is an intermediate in the synthesis of styrene, which is used to make plastics and elastomers. Cumene (22%) is used to produce phenol and acetone. Phenols are used in the manufacture of phenolic resins and nylon intermediates; acetone is used as a solvent and in the manufacture of pharmaceuticals. Cyclohexane (15%) is used to make nylon resins. Other industrial chemicals manufactured from benzene include nitrobenzene (7%), which is used in the production of aniline and other products, urethanes, linear alkylbenzenes used for detergents (2%), chlorobenzenes for engineering polymers (1%), and miscellaneous other uses (1%) (Eveleth 1990; Greek 1990; IARC 2018; NLM 2023).

Benzene is also a component of gasoline since it occurs naturally in crude oil and since it is a byproduct of oil refining processes (Brief et al. 1980; Holmberg and Lundberg 1985). Benzene is especially important for unleaded gasoline because of its anti-knock characteristics. Historically, the percentage by volume of benzene in unleaded gasoline previously was approximately 1–2% (NESCAUM 1989); however, beginning in 2011, the annual average benzene content of gasoline is regulated to 0.62% volume (maximum average: 1.3% volume) under the Mobile Source Air Toxics rules (EPA 2023a). Benzene is present at insignificant levels in diesel fuel (EPA 1993).

The widespread use of benzene as a solvent has decreased. This is likely the result of EPA listing benzene as a hazardous air pollutant (HAP), hazardous waste (EPA 1977, 1981), and human carcinogen (IRIS 2003). Many products that used benzene as solvents in the past have replaced it with other organic solvents; however, benzene may still occur as a trace impurity in these products (Kim et al. 2022). Less than 2% of the amount produced is used as a solvent in products such as trade and industrial paints, rubber cements, adhesives, paint removers, artificial leather, and rubber goods.

In the past, benzene has been used in the shoe manufacturing and rotogravure printing industries in the United States (EPA 1978; OSHA 1977) and continues to be used in the printing industry outside of the United States (Shi et al. 2022). Furthermore, small amounts of benzene were present in certain consumer products (such as some paint strippers, carburetor cleaners, denatured alcohol, and rubber cement used in tire patch kits and arts and crafts supplies) contained small amounts of benzene (Young et al. 1978). Other consumer products that contained benzene were certain types of carpet glue, textured carpet liquid detergent, and furniture wax (Wallace et al. 1987).

The Consumer Products Safety Commission (CPSC) withdrew an earlier proposal to ban consumer products, except gasoline and laboratory reagents, that contained benzene as an intentional ingredient or as a contaminant at >0.1% by volume. The withdrawal of the rulemaking was based on CPSC findings that benzene was no longer used as an intentional ingredient and that the contaminant levels remaining in certain consumer products were unlikely to result in significant exposures (NTP 1994). Products containing >5% benzene, and paint solvents and thinners containing <10% of petroleum distillates such as benzene, are required to meet established labeling requirements. In a guidance document targeting school science laboratories, the CPSC recommended that benzene not be used or stored in schools. The document identified benzene as a carcinogen and ascertained that the hazards posed by its use in high school laboratories may be greater than its potential usefulness.

The U.S. Food and Drug Administration (FDA) regulates benzene as an indirect food additive under the Food, Drug, and Cosmetics Act (FDCA). Under the FDCA, benzene is restricted to use only as a component of adhesives used on articles intended for packaging, transport, or holding foods (FDA 1977a). The FDA has set a limit of <1 ppm on residual benzene in modified hop extract (where benzene is a solvent used in sequential extraction during production of this flavoring agent) used as an additive in beer (FDA 1977b). In December 2023, the FDA requested that the U.S. Pharmacopeia-National Formulary avoid use of carbomers manufactured with benzene, which are used as thickening agents in drug and cosmetic products, due to residual benzene levels in end-use products above the maximum permitted level of 2 ppm (FDA 2023a). The International Council for Harmonisation of Technical Requirements for Pharmaceuticals for Human Use (ICH) classified benzene as a “solvent to be avoided,” indicating that it should only be used if absolutely necessary during the manufacture of a drug with a “significant therapeutic advance” (ICH 2021). For these cases, ICH set a maximum permitted level of 2 ppm.

5.2.4 Disposal

Benzene-containing wastes, such as commercial chemical products, manufacturing chemical intermediates, and spent solvents, are subject to federal and/or state hazardous waste regulations. Waste byproducts from benzene production processes include acid and alkali sludges, liquid-solid slurries, and solids (EPA 1982; Saxton and Narkus-Kramer 1975). In the past, landfilling and lagooning have been the major methods of disposal of benzene-containing industrial wastes (EPA 1982). Biodegradation is the primary fate of industrial wastes; however, a portion of the benzene is expected to be lost due to volatilization. Unfortunately, benzene, along with other hazardous contaminants, also leaches into groundwater from the lagooned wastes.

The recommended method of disposal is to incinerate solvent mixtures and sludges at a temperature that ensures complete combustion. The recommended methods for combustion are liquid injection incineration at a temperature range of 650–1,600°C and a residence time of 0.1–2 seconds; rotary kiln incineration at a temperature range of 820–1,600°C and residence times of seconds for liquids and gases, and hours for solids; and fluidized bed incineration at a temperature range of 450–980°C and residence times of seconds for liquids and gases and longer for solids (IRPTC 1985). Since benzene burns with a very smoky flame, dilution with alcohol or acetone is suggested to minimize smoke.

Several methods exist for the treatment of wastewater that contains benzene: biological treatment (aeration or activated sludge process), solvent extraction, air and/or steam stripping, and activated carbon process (EPA 1994a; IRPTC 1985). A combination of steam stripping and air stripping, and a vapor extraction system that removes the separated benzene vapor may be suitable for the treatment of contaminated groundwater and soil (Naft 1992). An *in situ* bioremediation process has been used to decontaminate a site by delivering a controlled amount of nitrate (to accelerate biodegradation of benzene) to the site under hydraulic control (Kennedy and Hutchins 1992).

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 2022b). 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 2022b).

5.3.1 Air

Estimated releases of 3,198,877 pounds (~1,450 metric tons) of benzene to the atmosphere from 1,125 domestic manufacturing and processing facilities in 2022, accounted for about 36% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2023). These releases are summarized in Table 5-3.

Table 5-3. Releases to the Environment from Facilities that Produce, Process, or Use Benzene[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	17	96,461	1	0	3,790	19	96,478	3,792	100,270
AK	18	6,281	0	0	225	604	6,285	825	7,110
AZ	17	12,184	0	0	5	510	12,189	510	12,699
AR	8	39,584	1,087	10	74	1,168	40,748	1,175	41,923
CA	47	26,101	4,194	0	50,524	342	72,364	8,796	81,161
CO	26	34,104	3	26,828	908	6,781	34,105	34,519	68,625
CT	3	3,189	2	0	0	0	3,190	0	3,190
DE	1	2,190	9	0	5	0	2,199	5	2,204
FL	24	18,491	0	150	0	1	18,641	1	18,642
GA	7	2,111	1	0	0	0	2,112	0	2,112
HI	10	15,423	0	3	112	0	15,427	112	15,538
ID	3	854	0	0	17	0	854	17	871
IL	34	117,031	112	0	290	96	117,317	212	117,529
IN	33	92,888	16	16,867	1,053	103	109,758	1,169	110,927

Table 5-3. Releases to the Environment from Facilities that Produce, Process, or Use Benzene[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
IA	42	64,382	1	0	0	0	64,383	0	64,383
KS	22	35,280	18	172	1,530	960	36,613	1,347	37,960
KY	22	78,976	731	0	201	7,674	79,712	7,871	87,582
LA	75	430,040	543	6,057	803	773	435,923	2,294	438,216
ME	7	1,042	1	0	0	46	1,043	46	1,089
MD	6	960	5	0	0	0	965	0	965
MA	8	2,211	2	0	17	185	2,213	202	2,415
MI	30	128,407	26	0	133	1,366	128,552	1,380	129,932
MN	19	14,884	31	0	14	2	14,915	16	14,931
MS	14	38,867	5	70,056	175	2	38,872	70,233	109,105
MO	14	31,700	0	0	0	169	31,700	169	31,869
MT	9	20,264	6	0	66	11	20,318	30	20,348
NE	21	4,920	0	0	75	1,370	4,920	1,445	6,365
NV	3	396	0	0	0	3	396	3	399
NH	5	0	0	0	0	0	0	0	0
NJ	10	11,816	310	0	6	17	12,126	23	12,149
NM	24	65,825	0	2,706,450	1,078	5,130	2,754,512	23,971	2,778,483
NY	17	5,158	1	0	7	1	5,162	4	5,167
NC	10	1,983	0	0	4	195	1,987	195	2,182
ND	29	28,432	2	17,856	11	0	28,435	17,866	46,302
OH	36	59,324	7	6,564	72	664	59,347	7,284	66,631
OK	31	67,899	28	2,012	14,590	21	67,926	16,623	84,549
OR	4	1,101	2	0	3	0	1,102	4	1,106
PA	21	90,869	143	16	118	1,123	91,011	1,258	92,269
RI	3	853	1	0	0	0	854	0	854
SC	5	60,063	0	0	0	0	60,063	0	60,064
SD	16	6,343	0	0	0	0	6,343	0	6,343
TN	17	17,936	155	0	263	0	17,937	418	18,355
TX	260	1,082,656	2,465	2,528,552	11,719	4,596	1,679,417	1,950,570	3,629,987
UT	15	132,709	1,027	14,564	163	128	132,721	15,870	148,591
VA	6	7,333	1,426	0	0	108	8,713	153	8,867
WA	16	33,109	8	0	222	59	33,168	230	33,398
WV	11	13,850	1	8,115	371	4	14,205	8,136	22,341
WI	18	139,633	0	0	764	3	139,633	767	140,399
WY	17	21,992	0	16,797	92,649	0	22,508	108,930	131,438
GU	2	1,397	0	0	0	0	1,397	0	1,397

Table 5-3. Releases to the Environment from Facilities that Produce, Process, or Use Benzene[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
MP	2	208	0	0	0	0	208	0	208
PR	7	6,198	0	0	0	0	6,198	0	6,198
VI	3	22,970	0	0	0	0	22,970	0	22,970
Total	1,125	3,198,877	12,369	5,421,070	182,059	34,235	6,560,134	2,288,475	8,848,609

[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 2023 (Data are from 2022)

Benzene is released into the atmosphere from both natural and industrial sources. Natural sources include, but are not limited to, crude oil seeps, forest fires, volcanoes, and plant volatiles (Brief et al. 1980; Dickinson et al. 2022; Graedel 1978; NCI 2022). Major anthropogenic sources of benzene include industrial emissions, automobile exhaust, automobile refueling operations, and environmental tobacco smoke.

Industrial and automotive sources of benzene are well characterized (Tables 5-2 and 5-3). Oil and gas production is expected to be the largest industrial contributor to benzene emissions: around 20% of total emissions (EPA 2020a). Benzene composes 3–5% of passenger car tailpipe emissions, depending on the control technology and fuel, and is estimated to be about 1% of evaporative emissions from vehicles (EPA 1993). Mobile sources (including on-road vehicles) were estimated to contribute to 33% of all benzene emissions, and on-road vehicles were estimated to contribute around 12% of all benzene emissions (EPA 2020a).

Emissions from car exhaust began decreasing in localities where federally reformulated gasoline is sold as part of a mandate under the 1990 Clean Air Act (EPA 1995b). This program has had great success, reducing benzene emissions by 43% between 1995 to 1999 in the District of Columbia and the 17 participating states (EPA 1999). Gasoline-related benzene emissions have further been reduced under the mobile source air toxics rule (EPA 2007). These new standards are expected to reduce benzene emissions by 20,000 per year by 2030 (EPA 2017).

Benzene is also released by off-gassing from particle board (Glass et al. 1986), vaporization from oil spills, and emissions from landfills (Bennett 1987; Wood and Porter 1987). An analysis of gas from 20 Class II (municipal) landfills revealed a maximum concentration of 32 ppm for benzene (Wood and Porter 1987). While all of these sources release more benzene into the environment, a large percentage of the benzene inhaled by humans comes from tobacco cigarette smoke. Exhaled breath of smokers contains benzene (Wallace 1989a, 1989b; Wallace and Pellizzari 1986; Wester et al. 1986).

Fires are the natural sources most monitored for benzene releases to the air (Austin et al. 2001; Lowry et al. 1985). Austin et al. (2001) monitored volatile organic compounds (VOCs) released from nine municipal fires in Canada and found a mean concentration of 3.45 ppm of benzene. This study also reported very high relative concentration of other VOCs.

EPA's National Emission Inventory (NEI) database contains information regarding sources that emit criteria air pollutants (CAPs) and their precursors, and 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. Benzene emissions estimated from the 2020 inventory are summarized in Table 5-4.

Table 5-4. National Emission Inventory (NEI) Total National Emissions for Benzene Estimated by Sector 2020

Sector	Emissions (pounds)
Fires; wildfires	80,674,309
Industrial processes; oil and gas production	65,225,185
Mobile; non-road equipment; gasoline	49,281,476
Fires; prescribed fires	39,151,086
Mobile; on-road non-diesel light duty vehicles	36,787,086

Table 5-4. National Emission Inventory (NEI) Total National Emissions for Benzene Estimated by Sector 2020

Sector	Emissions (pounds)
Fuel combustion; residential; wood	27,232,070
Waste disposal	9,722,888
Mobile; non-road equipment; diesel	4,142,396
Fires; agricultural field burning	3,063,972
Gas stations	2,435,276
Industrial processes; storage and transfer	2,018,389
Mobile; aircraft	1,361,523
Mobile; on-road non-diesel heavy duty vehicles	1,286,764
Fuel combustion; industrial boilers, internal combustion engines; biomass	1,048,989
Industrial processes; chemical manufacturing	990,792
Mobile; on-road diesel heavy duty vehicles	943,279
Agriculture; livestock waste	905,710
Mobile; locomotives	895,965
Fuel combustion; industrial boilers, internal combustion engines; natural gas	842,202
Bulk gasoline terminals	818,626
Miscellaneous non-industrial not elsewhere classified	800,091
Industrial processes; petroleum refineries	619,480
Solvent; consumer and commercial solvent use	593,762
Industrial processes; not elsewhere classified	504,294
Industrial processes; cement manufacturing	447,863
Commercial cooking	384,291
Mobile; on-road diesel light duty vehicles	379,016
Industrial processes; ferrous metals	337,533
Fuel combustion; electric generation; coal	178,053
Fuel combustion; electric generation; biomass	172,574
Solvent; industrial surface coating and solvent use	169,271
Industrial processes; pulp and paper	158,119
Fuel combustion; electric generation; natural gas	105,881
Mobile; commercial marine vessels	89,440
Fuel combustion; electric generation; oil	89,325
Fuel combustion; industrial boilers, internal combustion engines; other	84,235
Fuel combustion; commercial/institutional; biomass	84,118
Mobile; non-road equipment; other	60,133
Fuel combustion; industrial boilers, ices; oil	53,089
Industrial processes; non-ferrous metals	29,347
Fuel combustion; commercial/institutional; natural gas	28,728
Fuel combustion; commercial/institutional; oil	28,534
Fuel combustion; industrial boilers, internal combustion engines; coal	12,801
Fuel combustion; electric generation; other	11,345

Table 5-4. National Emission Inventory (NEI) Total National Emissions for Benzene Estimated by Sector 2020

Sector	Emissions (pounds)
Fuel combustion; residential; natural gas	10,061
Fuel combustion	3,839
Industrial processes; mining	2,960
Fuel combustion; residential; other	854
Solvent; graphic arts	634
Fuel combustion; residential; oil	597
Solvent; degreasing	477
Fuel combustion; comm/institutional; coal	175
Solvent; dry cleaning	2

Source: EPA 2020a

5.3.2 Water

Estimated releases of 12,369 pounds (~5.61 metric tons) of benzene to surface water from 1,125 domestic manufacturing and processing facilities in 2022, accounted for about 0.14% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2023). This estimate includes releases to wastewater treatment and publicly owned treatment works (POTWs) (TRI22 2023). These releases are summarized in Table 5-3.

Benzene is released to water from the discharges of both treated and untreated industrial wastewater, gasoline leaks from underground storage tanks, accidental spills during transportation of chemical products or drilling, and leachate from landfills and other contaminated soils (CDC 1994; Crawford et al. 1995; EPA 1979; NESCAUM 1989; Staples et al. 1985; Reddy et al. 2012). A fire in a tire dump site in western Frederick County, Virginia, produced a free-flowing oily tar containing benzene among other chemicals. The seepage from this site contaminated nearby surface water (EPA 1992). Accidental spills released 3,000 gallons of benzene into Newark Bay and its major tributaries in 1991 (Crawford et al. 1995). Following a tanker truck accident that released 3,200 gallons of gasoline in 2020, benzene was detected in the Yellowstone River and monitoring wells (EPA 2020b). Oil spills are an important source of emissions to the marine environment. An estimated 1,600 metric tons of benzene were released from the 2010 Deepwater Horizon oil spill in the Gulf of Mexico (Reddy et al. 2012). In 2020, a tanker truck accident resulted in approximately 3,200 gallons of gasoline spilled at Yellowstone National Park; soil monitoring was not reported, but increased benzene was detected in the nearby river and monitoring wells (EPA 2020b).

5.3.3 Soil

Estimated releases of 182,059 pounds (~82.6 metric tons) of benzene to soil from 1,125 domestic manufacturing and processing facilities in 2022, accounted for about 2.1% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2023). An additional 5,421,070 pounds (~2,460 metric tons), constituting about 61% of the total environmental emissions, were released via underground injection (TRI22 2023). These releases are summarized in Table 5-3.

Benzene is released to soils through industrial discharges, land disposal of benzene-containing wastes, and gasoline leaks from underground storage tanks. Limited data on terrestrial emissions are available. This may be due to benzene's expected partitioning to air or migration to groundwater. Benzene was detected in sediment at an NPL site of a former truck terminal, which disposed of wastewater in an unlined lagoon and in underground tanks (ATSDR 2023d), and in the soil of the marine terminal area of a former army air and ground base (ATSDR 2019a). In northern Virginia, approximately 200,000 gallons of liquid hydrocarbons were released from a fuel-storage terminal into the underlying soil (Mushrush et al. 1994).

5.4 ENVIRONMENTAL FATE

5.4.1 Transport and Partitioning

Air. The high volatility of benzene is the controlling physical property in the environmental transport and partitioning of this chemical. Benzene is considered to be highly volatile with a vapor pressure of 94.8 mm Hg at 25°C. Benzene is soluble in water, with a solubility of 1,790 mg/L at 25°C, and the Henry's law constant for benzene (5.5×10^{-3} atm-m^3/mole at 25°C) indicates that benzene partitions readily to the atmosphere from surface water (Mackay and Leinonen 1975; NLM 2023). Since benzene is soluble in water, removal from the atmosphere via wet deposition may occur. A substantial portion of any benzene in rainwater that is deposited to soil or water will be returned to the atmosphere via volatilization.

Water. Benzene is soluble in water and (1,790 mg/L at 25°C) and has low tendency to partition to the organic phase based on the octanol/water partition coefficient (log K_{ow}) of 2.13 (NLM 2023). Due to this and its volatility, benzene will preferentially partition to the atmosphere rather than be removed from the water column through sorption to particulate matter or sediments.

Sediment and Soil. Benzene released to soil surfaces partitions to the atmosphere through volatilization, to surface water through runoff, and to groundwater as a result of leaching. The soil organic carbon sorption coefficient (K_{oc}) for benzene has been measured with a range of 60–85 (Karickhoff 1981; Kenaga 1980; NLM 2023), indicating that benzene is highly mobile in soil and readily leaches into groundwater. Other parameters that influence leaching potential include the soil type (e.g., sand versus clay), amount of rainfall, depth of the groundwater, and extent of degradation. In a study of the sorptive characteristics of benzene to groundwater aquifer solids, benzene showed a tendency to adsorb to aquifer solids. Greater soil adsorption was observed within an aquifer with more organic matter content (4.4%), compared to an aquifer with less organic matter content (2.2%) (Uchrin and Mangels 1987). An investigation of the mechanisms governing the rates of adsorption and desorption of benzene by dry soil grains revealed that periods of hours are required to achieve equilibrium and that adsorption is much faster than desorption (Lin et al. 1994). The rate of volatilization and leaching are the principal factors that determine overall persistence of benzene in sandy soils (Tucker et al. 1986).

Other Media. Studies suggest that benzene does not bioaccumulate in marine organisms. The bioconcentration/ bioaccumulation potential of benzene in aquatic organisms of the open coastal ocean was investigated by sampling final effluent from the Los Angeles County wastewater treatment plant quarterly from November 1980 to August 1981 (Gossett et al. 1983). Benzene has a relatively low log K_{ow} value of 2.13 (Gossett et al. 1983; NLM 2023). In the alga, *Chlorella,* a bioaccumulation factor of 30 was determined experimentally (Geyer et al. 1984). An experimental bioconcentration factor (BCF) of 4.27 (reported as log BCF of 0.63) was measured in goldfish reared in water containing 1 ppm of benzene (Ogata et al. 1984). Based on these measured values, bioconcentration/bioaccumulation of benzene in the aquatic food chains does not appear to be an important fate pathway. These results are consistent with the fact that benzene has a relatively low octanol/water partition coefficient (Gossett et al. 1983; NLM 2023), suggesting relatively low bioaccumulation. There is no evidence in the literature of biomagnification of benzene in aquatic food chains.

Evidence exists for the uptake of benzene by cress and barley plants from soil (Scheunert et al. 1985; Topp et al. 1989). BCFs for barley plants after 12, 33, and 125 days were 17, 2.3, and 4.6, respectively.

BCFs for cress plants after 12, 33, and 79 days were 10, 2.3, and 1.9, respectively. The relative decrease in the BCFs with time was attributed to growth dilution (Topp et al. 1989). Since benzene exists primarily in the vapor phase, air-to-leaf transfer is considered to be the major pathway of vegetative contamination (Hattemer-Frey et al. 1990). Based on an equation to estimate vegetative contamination, the total concentration of benzene on exposed food crops consumed by humans and used as forage by animals was estimated to be 587 ng/kg, 81% of which was from air-to-leaf transfer and 19% was from root uptake (Hattemer-Frey et al. 1990).

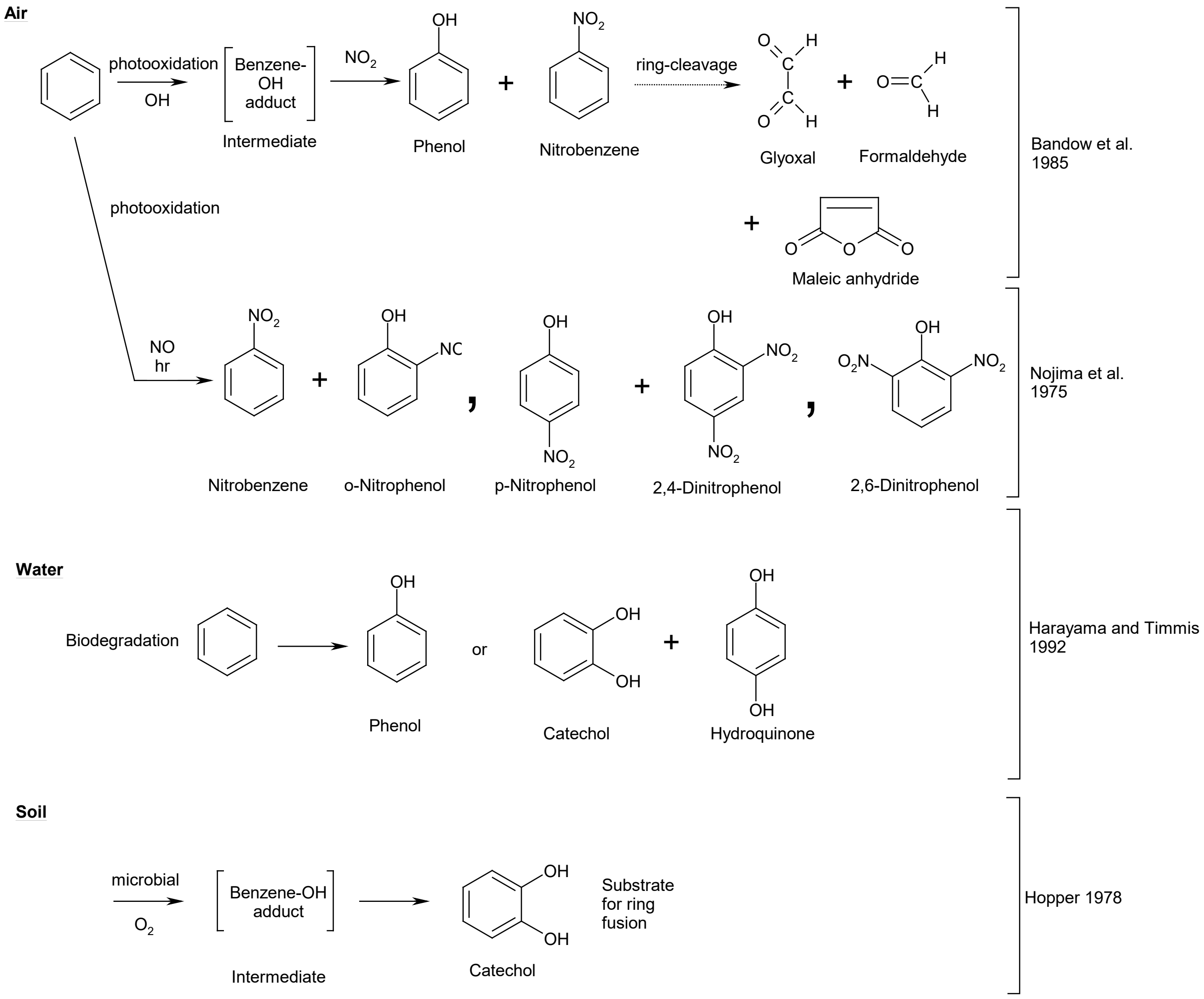
Benzene also accumulates in the leaves and fruits of plants. After 40 days, plants grown in benzene-rich environments showed bioaccumulation in the leaves and fruit that were greater than the air portioning coefficient of benzene in the atmosphere. Blackberries exposed to 0.313 ppm and apples exposed to 2.75 ppm contained about 1,000 and 36 ng/g of benzene, respectively (Collins et al. 2000).

5.4.2 Transformation and Degradation

Benzene undergoes a number of different transformation and degradation reactions in the environment as discussed in the following sections. The resulting environmental transformation products within different media are shown in Figure 5-2.

Air. Benzene in the atmosphere exists predominantly in the vapor phase (Eisenreich et al. 1981). The most important degradation process for benzene is its reaction with atmospheric hydroxyl radicals. The rate constant for the vapor phase reaction of benzene with photochemically produced hydroxyl radicals has been determined to be 1.3×10^{-12} cm^3/molecule-second (Gaffney and Levine 1979), which corresponds to a calculated residence time of 8 days at an atmospheric hydroxyl radical concentration of 1.1×10^6 molecules/cm^3, based on the equation reported by Lyman (1982). Using a hydroxyl radical concentration of 1×10^8 molecules/cm^3, corresponding to a polluted atmosphere (Lyman 1982), the calculated residence time is shortened to 2.1 hours. Benzene may also react with other oxidants in the atmosphere such as nitrate radicals and ozone; however, the rate of degradation is considered insignificant compared to the rate of reaction with hydroxyl radicals. Residence times of 472 years for rural atmospheres and 152 years for urban atmospheres were calculated for the reaction of benzene with ozone (O_3) using a rate constant of 7×10^{-23} cm^3/molecule-second (Pate et al. 1976) and atmospheric concentrations of 9.6×10^{11} O_3 molecules/cm^3 (rural) and 3×10^{12} O_3 molecules/cm^3 (urban) (Lyman 1982).

Figure 5-2. Environmental Transformation Products of Benzene in Various Media



Sources: Bandow et al. 1985; Harayama and Timmis 1992; Hopper 1978; Nojima et al. 1975

The reaction of benzene and nitric oxide in a smog chamber was investigated to determine the role of benzene in photochemical smog formation (Levy 1973). The results showed that benzene exhibited low photochemical smog reactivity in the four categories tested: rate of photooxidation of nitric oxide, maximum oxidant produced, eye-irritation response time, and formaldehyde formation. The study authors concluded that benzene probably does not play a significant role in photochemical smog formation (Levy 1973). In the presence of active species such as nitrogen oxides and sulfur dioxide, the rate of photodegradation of benzene in the gas phase was greater than that in air alone. Its half-life in the presence of such active species (100 ppm benzene in the presence of 10–110 ppm NO_x or 10–100 ppm SO_2) was 4–6 hours, with 50% mineralization to carbon dioxide in approximately 2 days (Korte and Klein 1982). The primary products of the reaction of benzene with nitrogen monoxide gas include nitrobenzene, *o*- and *p*-nitrophenol, and 2,4- and 2,6-dinitrophenol (Nojima et al. 1975). Photooxidation of benzene in a nitrogen monoxide/nitrogen dioxide-air system formed formaldehyde, formic acid, maleic anhydride, phenol, nitrobenzene, and glyoxal (ethane-1,2-dione) (Bandow et al. 1985).

Direct photolysis of benzene in the atmosphere is not likely because the upper atmosphere effectively filters out wavelengths of light <290 nm, and benzene does not absorb wavelengths of light >260 nm (Bryce-Smith and Gilbert 1976).

Water. Benzene is subject to indirect photolysis in sunlit surface water but does not undergo direct photolysis. For direct photolysis to occur, a substance must absorb photons of light >290 nm. During indirect photolysis, light energy is absorbed by other constituents (photosensitizers) of the media (water, soil) and the excited species can then transfer energy to benzene (indirectly promoting it to an excited electronic state), or lead to the formation of reactive species, such as singlet oxygen or hydroxy radicals, which react with benzene. Humic and fulvic acids are well-known photosensitizing agents and are practically ubiquitous in natural waters. A half-life of 16.9 days was reported for photolysis of benzene dissolved in oxygen-saturated deionized water and exposed to sunlight (Hustert et al. 1981).

Benzene is readily degraded in water under aerobic conditions. Results of a biochemical oxygen demand (BOD) test determined that benzene was completely biodegradable after the second week of static incubation at 25°C at benzene concentrations of 5 and 10 mg/L using domestic wastewater as the microbial inoculum (Tabak et al. 1981). A study of the degradation of benzene by the microbial population of industrial wastewater at 23°C using a shaker flask system showed that after 6 hours, only 8% (4 mg/L) of the initial 50 mg/L dose of benzene remained (Davis et al. 1981). Water from a petroleum production site was successfully biotreated for complete removal of benzene using a flocculated culture of *Thiobacillus denitrificans* strain F and mixed heterotrophs (Rajganesh et al. 1995).

Microbial degradation of benzene in aquatic environments is influenced by many factors including microbial population, dissolved oxygen, nutrients, other sources of carbon, inhibitors, temperature, pH, and initial concentration of benzene. Vaishnav and Babeu (1987) reported biodegradation half-lives for benzene in surface water (river water) and groundwater of 16 and 28 days, respectively. Benzene was found to be resistant to biodegradation in surface water taken from a harbor and supplemented with either nutrients (nitrogen and phosphorus) or acclimated microbes; however, biodegradation did occur, with a half-life of 8 days, in surface water enriched with both nutrients and microbes (Vaishnav and Babeu 1987). At very high levels, as may be the case of a petroleum spill, benzene (and other compounds contained in petroleum) is toxic to microorganisms and the rate of degradation is slow compared to low initial starting concentrations. In another study, Davis et al. (1994) observed rapid aerobic biodegradation of benzene in aquifer groundwater samples and measured times for 50% disappearance ranging from 4 days for an initial benzene concentration of 1 mg/kg to 14 days for an initial benzene concentration of 10 mg/kg. Under acidic conditions (pH 5.3, 20°C), benzene was completely microbially degraded in 16 days in groundwater taken from a shallow well (Delfino and Miles 1985).

The aerobic biodegradation of benzene is also influenced by the presence of other aromatic hydrocarbons. A bacterial culture grown with aromatic hydrocarbons plus nitrogen-, sulfur-, and oxygen-containing aromatic compounds was much less efficient in degrading benzene than the culture grown with aromatic hydrocarbons alone. Pyrrole strongly inhibited benzene degradation. Benzene degradation was high when either toluene or xylene were present (Arvin et al. 1989).

Laboratory studies on microbial degradation of benzene with mixed cultures of microorganisms in gasoline-contaminated groundwater revealed that both oxygen and nitrogen concentrations are major controlling factors in the biodegradation of benzene. Nitrogen enhanced the biodegradation rate of benzene 4.5-fold, over inoculum-enriched water alone. More than 95% of the benzene in groundwater was removed through microbial action within 73.5 hours (Karlson and Frankenberger 1989).

Benzene biodegradation under anaerobic conditions does not readily occur. When dissolved oxygen is depleted, an alternative electron acceptor such as nitrate, carbonate, or iron (III) must be available, and microbes capable of using the alternative electron acceptor to degrade the benzene must be present (McAllister and Chiang 1994). Using aquifer material obtained from a landfill from Norman, Oklahoma, no significant benzene biodegradation was reported during the first 20 weeks of incubation under anaerobic conditions at 17°C; however, after 40 weeks of incubation, benzene concentrations were reduced by 72 and >99% of the benzene was degraded after 120 weeks (Wilson et al. 1986). No degradation of benzene was observed in 96 days under anaerobic conditions (20°C) using raw water intake from a water treatment plant (Delfino and Miles 1985).

Use of water as an oxygen source in the anaerobic degradation of benzene has been demonstrated. Experiments indicated that incorporation of ^{18}O from ^{18}O-labeled water is the initial step in the anaerobic oxidation of benzene by acclimated methanogenic cultures. Phenol was the first major product (Vogel and Grbić-Galić 1986).

Sediment and Soil. Benzene is biodegraded in soil under aerobic conditions. This process has been well-described, as reviewed by Gibson (1980, 1977) and Hopper (1978). Microbial metabolism of benzene proceeds through the formation of *cis*-dihydrodiols and, with further metabolism, to catechols, which are the substrates for ring fission. *Pseudomonas putida* oxidized benzene through *cis*-1,2-dihydroxy-1,2-dihydrobenzene (Gibson 1977; Hopper 1978). A strain of *Rhodococcus* isolated from contaminated river sediment mineralized 71% of benzene at an initial concentration of 0.7 mg/L in 14 days (Malachowsky et al. 1994). The soil bacterium, *Nitrosomonas europaea*, catabolized benzene to phenol and hydroquinone (Keener and Arp 1994). Another mixotrophic bacteria, a strain of *Pseudomonas sp.* isolated from contaminated soil, grew under both anaerobic and aerobic conditions and used benzene for its growth (Morikawa and Imanaka 1993). Furthermore, *Norcardia sp*. and *Pseudomonas sp.* effectively degraded benzene to carbon dioxide within 7 days (45–90%) (Haider et al. 1981). The biodegradation of 2 mg of radiolabeled benzene in 100 g of soil with a mixed microbial population transformed 47% of the added radioactivity to carbon dioxide after 10 weeks (Haider et al. 1981). The study authors concluded that specific organisms that degrade benzene were present in the soil in only small numbers.

Limited benzene degradation was predicted in shallow sandy soil contaminated with gasoline from an leaking underground storage tank using ULTRA, a fate and transport model used to predict the environmental fate (Tucker et al. 1986). The ULTRA model predicted that only about 1% of the benzene in the gasoline would be degraded over a 17-month period, and 3% would remain in the soil (Tucker et al. 1986). Most of the benzene present in the soil from leaked gasoline was predicted to either volatilize (67%) or move into groundwater (29%).

Salanitro (1993) summarized the aerobic degradation rates for BTEX in laboratory subsoil-groundwater slurries and aquifers. Decay rates for benzene in laboratory microcosms were highest (19–52% per day) for benzene concentrations <1 ppm when initial dissolved oxygen levels were about 8 ppm. Rates were significantly reduced (0–1.1% per day) when benzene levels were 1–2 ppm, and no degradation was observed when benzene levels were >2 ppm. This is particularly relevant in the case of petroleum spills as high concentrations of petroleum compounds are toxic to organisms and decrease the rate of biodegradation.

Benzene has been shown to be anaerobically transformed by mixed methanogenic cultures derived from ferulic acid-degrading sewage sludge enrichments. In most of the experiments, benzene was the only semi-continuously supplied energy sources in the defined mineral medium (Grbić-Galić and Vogel 1987).

After an initial acclimation time of 11 days, at least 50% of the substrate was converted to CO_2 and methane. The intermediates were consistent with benzene degradation via initial oxidation by ring hydroxylation.

Edwards and Grbić-Galić (1992) discussed that where mixtures of benzene, toluene, xylenes, and ethylbenzene are present in an anaerobic environment, there is a sequential utilization of the substrate hydrocarbons, with toluene usually being the first to be degraded, followed by xylene isomers, then finally ethylbenzene and benzene. They proposed that benzene may not be degraded at all. However, in microcosm experiments, Edwards and Grbić-Galić (1992) demonstrated that benzene at initial concentrations of 40–200 μM degraded at rates ranging from 0.36 to 3.7 μM/day, depending upon substrate concentration and the presence of other carbon sources.

Other Media. Twenty-day-old spinach leaves placed in a hermetic chamber containing vapors of ^{14}C-labeled benzene were shown to assimilate benzene. The benzene was subsequently metabolized to various nonvolatile organic acids (Ugrekhelidze et al. 1997).

Bacterium strain, *Mycobacterium cosmeticum* byf-4, has been reported to aerobically biodegrade benzene, toluene, ethylbenzene, and *o*-xylene, simultaneously or individually, via mineralization and incorporation into cell materials (Zhang et al. 2013). Benzene alone or in a mixture of benzene, toluene, ethylbenzene, and *o*-xylene, at an initial concentration of 100 mg/L, was completely degraded within 36–42 hours.

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to benzene depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of benzene 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 benzene 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 5-5 shows the lowest limit of detections that are achieved by analytical analysis in environmental media. An overview summary of the range of concentrations detected in environmental media is presented in Table 5-6.

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

Media	Detection limit	Reference
Air (ppbv)	0.002	EPA 2023b
Drinking water (ppb)	0.028	EPA 2013
Surface water and groundwater (ppb)	0.0067	USGS 2003
Soil (ppb)	0.009	EPA 1996
Sediment (ppb)	0.009	EPA 1996
Whole blood (ppb)	0.024	CDC 2022a

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

Table 5-6. Summary of Environmental Levels of Benzene

Media	Low	High[a]	For more information
Outdoor air (ppbv)	0.082	46.6	Section 5.5.1
Indoor air (ppbv)	<0.02	140	Section 5.5.1
Surface water (ppb)	0.038[b]	7.8	Section 5.5.2
Groundwater (ppb)	0.01	27,000	Section 5.5.2
Drinking water (ppb)	0.02	923	Section 5.5.2
Soil (ppb) (2023 data)	ND	0.026	section 5.5.3
Food (ppb)	ND	190	section 5.5.4

[a] High levels may be representative of monitoring data at localized contaminated sites and may not be reflective of background environmental levels.
[b] Surface water data were limited; this value represents the average level detected in 2023. The lowest level detected was not reported.

ND = not detected (and detection limit not specified)

Detections of benzene in air, water, and soil at NPL sites are summarized in Table 5-7.

Table 5-7. Benzene 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)	34	82.5	1.25	2	432
Soil (ppb)	1,890	1,440	75.3	261	183

Table 5-7. Benzene 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
Air (ppbv)	3.72	9.11	25.1	224	139

[a]Concentrations found in ATSDR site documents from 1981 to 2022 for 1,868 NPL sites (ATSDR 2022a). 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

Benzene is ubiquitous in the atmosphere. It has been identified in outdoor air samples of both rural and urban environments and in indoor air, and concentrations vary seasonally (Kinney et al. 2002). However, ambient air concentrations, particularly in urban environments, have decreased in the past few decades.

In California, motor vehicle exhaust accounted for over 70% of the nonsmoking population's exposure to ambient benzene (Cal EPA 1987). The 1984 population-weighted average benzene concentration in California was estimated to be 3.3 ppbv (Cal EPA 1987). Benzene emissions in a Los Angeles roadway tunnel were measured at a concentration of 382 mg/L (118,420 ppmv) (Fraser et al. 1998). New regulations on benzene content in gasoline have dramatically reduced these levels (EPA 1995b, 2007, 2023a). The California EPA estimated that gasoline-attributed benzene decreased 70–80% between 1996 and 2014, and an updated statewide population-weighted average benzene concentration of approximately 0.25 ppbv was estimated (Cal EPA 2018).

Nationally, a decline in ambient air benzene of 66% has been reported between 1996 and 2009 (EPA 2014). Table 5-8 provides a comparison of benzene levels in outdoor air in various cities in the United States from data collected prior to implementation of national regulations reducing benzene content in gasoline, which were identified as primarily influenced by mobile sources (EPA 1987), to more recent data. This is a broad historical comparison for these urban areas; specific monitoring site locations and site types between the studies were not compared.

Table 5-8. Historical Comparison of Benzene Levels in Urban Air

Location	Year	Concentration (ppbv)	Reference
San Francisco, California	1984	0.8–5.2 (range); 2.6±1.3[a]	Wester et al. 1986
	2020–2023[b]	NS–1.39 (range); 0.110[d]	EPA 2023b[c]
St. Louis, Missouri	Summer 1985	0.63–12.1 (range); 1.85[d]	EPA 1987[c]
	2020–2023[b]	NS–0.52 (range); 0.17[d]	EPA 2023b[c]
Philadelphia, Pennsylvania	Summer 1985	0.32–3.0 (range); 1[d]	EPA 1987[c]
	2020–2023[b]	NS–9.43 (range); 0.18[d]	EPA 2023b[c]
New York (Manhattan), New York	Summer 1986	0.88–5.3 (range); 1.75[d]	EPA 1987[c]
	2020–2023[b]	NS–0.62 (range); 0.20[d]	EPA 2023b[c]
Boston, Massachusetts	1990–1991	0.69–3.1; 1.06[d]	Kelly et al. 1993
	2020–2023[b]	NS–0.37; 0.14[d]	EPA 2023b[c]
New York (Staten Island), New York	Spring 1984	0.1–34 (range); 4.4±6.6[a]	Singh et al. 1985
	2020–2023[b]	NS–0.93 (range); 0.20[d]	EPA 2023b[c]

[a] Average±standard deviation.
[b] As of October 26, 2023.
[c] Values were originally reported in parts per billion carbon (ppbC) and have been converted to ppbv benzene. ppbv = $\text{ppbC}_{\text{benzene}}$/six carbons.
[d] Median.

NS = not stated

Benzene is a pollutant monitored for in the national Air Quality System (AQS) database, which contains ambient air pollution data collected by EPA, state, local, and tribal air pollution control agencies from monitors throughout the country. Table 5-9 shows the yearly mean 24-hour percentile distributions of benzene at monitoring stations across the United States from 2019 to 2023. The maximum observed concentration for this time period (46.6 ppbv) was recorded from an industrial monitor in Detroit, Michigan in 2020. This value appears to be an outlier, as it was twice as high as the next highest monitoring concentration recorded in 2020, and the maximum concentration observed for the Detroit site in 2021 was of similar magnitude to other sites across the United States (data not shown).

Table 5-9. Summary of Annual Concentrations of Benzene (ppbv) Measured in Ambient Air at Locations Across the United States[a,b]

Year	Number of monitoring locations	Number of samples	Average	10th Percentile	50th Percentile	75th Percentile	95th Percentile	Maximum
2019	136	7,610	0.197	0.083	0.156	0.223	0.494	11.4
2020	172	9,237	0.217	0.082	0.166	0.254	0.485	46.6
2021	178	12,534	0.181	0.083	0.161	0.228	0.386	3.83

Table 5-9. Summary of Annual Concentrations of Benzene (ppbv) Measured in Ambient Air at Locations Across the United States[a,b]

Year	Number of monitoring locations	Number of samples	Average	10th Percentile	50th Percentile	75th Percentile	95th Percentile	Maximum
2022	131	3,291	0.177	0.087	0.155	0.221	0.394	1.68
2023[c]	117	2,613	0.176	0.099	0.158	0.211	0.363	3.01

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

Source: EPA 2023b

Dickinson et al. (2022) reported benzene air concentrations from wildfires within a 0–61-mile radius of the fire's origin. In 2019, benzene was detected in the air at mean concentrations of 3.968±6.287 ppbv (range: 0.042–25.000 ppbv) for the Idaho Nethkar wildfire and 0.446±0.167 ppbv (range: 0.165–0.668 ppbv) for the Washington state Williams Flats wildfire. In 2020, benzene air levels were measured at mean concentrations of 0.283±0.181 ppbv (range: 0.024–0.596 ppbv) for the Washington state Chief Timonthy wildfire and 1.826±1.781 ppbv (range: 0.104–4.000 ppbv) for the Idaho Whitetail Loop wildfire.

Benzene has been detected in indoor air. EPA's compilation of 14 studies of background indoor air concentrations found a 31–100% detection rate for benzene in 2,615 U.S. resident samples between 1990 and 2005 (EPA 2011). The background medians ranged from below the reporting level (0.05–1.6 μg/m^3; 0.02–0.50 ppbv) to 4.7 μg/m^3 (1.5 ppbv), with 95th percentiles of 9.9 to 29 μg/m^3 (3.1–9.2 ppbv), and maximum values between 21 and 460 μg/m^3 (6.6–140 ppbv). During the Detroit Exposure and Aerosol Research Study, which measured daily average ambient air concentrations of benzene from 2004 to 2007 in Detroit, Michigan, the mean daily ambient air concentrations were 1.3–4.1 μg/m^3 (0.4–1.3 ppbv; n=1,483) and the mean daily indoor air concentrations were 2.3–6.0 μg/m^3 (0.7–1.8 ppbv; n=934) (George et al. 2011).

Indoor air monitoring conducted in Detroit in November and December 2006, reported an average of 3±5.7 μg/m^3 benzene (0.9±1.8 ppbv) (Johnson et al. 2010). The ratio of indoor to outdoor air concentrations was 1.2, indicating that indoor air sources had a greater influence on the concentrations, likely a seasonal effect due to use of gas appliances and heaters and reduced ventilation. Smoking in homes was considered via questionnaire, but results differentiated by smoking versus nonsmoking homes were not reported. These levels were comparable to benzene levels of 3 µg/m^3 (0.9 ppbv) measured in the indoor air of a newly built home (Sasahara et al. 2007).

Between December 2003 and April 2006, benzene was detected at an average of 4.07±5.94 µg/m^3 (1.27±1.86 ppbv; 76% detection rate; range: <0.64–42 µg/m^3 or 0.20–13 ppbv) in New Jersey suburban and rural homes (Weisel et al. 2008). Few smokers were included in this study; the likely source of benzene was evaporative emissions from gasoline in cars housed in attached garages. Benzene was detected in the living rooms of two homes in Alaska between 1 and 25 ppbv, likely due to gasoline stored in garages outside of homes (Isbell et al. 2005). In a highly industrialized area of southeast Chicago, benzene was detected at an average of 4.1 µg/m^3 (1.3 ppbv) in homes (Van Winkle and Scheff 2001).

In the winter of 1999, mean benzene levels in New York City were 0.80±0.44 ppbv for outdoor (home) air, 1.8±2.2 ppbv for indoor (home) air, and 1.4±1.0 ppbv for personal air (Kinney et al. 2002). During the summer of 1999, mean benzene levels in New York City were 0.41±0.32 ppbv for outdoor (home) air, 0.53±0.27 ppbv for indoor (home) air, and 1.0±0.7 ppbv for personal air (Kinney et al. 2002). A review reported the benzene results of residential outdoor, indoor, and personal air monitoring from several studies conducted from the summer of 1999 to the spring of 2001 (Weisel 2010). Mean benzene values of 0.46±0.52, 0.84±1.3, and 0.93±1.4 ppbv for outdoor, indoor, and personal air, respectively, were reported for nonsmoking residences in Elizabeth, New Jersey. In Houston, Texas, the mean benzene values for nonsmoking households were 0.86±1.78, 1.6±2.0, and 1.6±1.6 ppbv for outdoor, indoor, and personal air, respectively. Additionally, nonsmoking residences in Los Angeles, California had mean benzene values of 0.84±0.78, 1.0±1.7, and 1.0±2.2 ppbv for outdoor, indoor, and personal air, respectively. The review did not discuss potential sources of indoor benzene pollution in-depth but noted that personal exposures were higher than indoor air exposures.

Residential heating oil is expected to be a source of benzene indoor air pollution. According to the New York Department of Health, the air from about 50% of oil fuel heated homes between the years 1997 and 2003 contained benzene concentrations ≥2.2 µg/m^3 (0.69 ppbv) inside the homes and 1.5 µg/m^3 (0.47 ppbv) in the area outside the homes (NYSDOH 2005). Concentrations in indoor air from 27 homes in Maine that heat with #2 fuel oil with basement tanks or with K1 with outside tanks were 0.39–13.3 µg/m^3 (0.12–4.16 ppbv) (Maine DEP 2014).

A significant source of indoor air benzene pollution is from residences with individuals who smoke tobacco inside the home. One study reported that a median level of benzene in 185 homes without smokers was 2.2 ppbv and a median level of benzene in 343 homes with one or more smokers was 3.3 ppbv (Wallace 1989a). This finding points to the possible significance of passive smoking as a source of benzene exposure. A study conducted by R.J. Reynolds Tobacco Company in smoking and nonsmoking homes revealed that benzene levels were elevated in smoking homes. In 24 nonsmoking homes, the mean benzene concentration was 1.21 ppbv, with a maximum of 5.93 ppbv. In 25 smoking homes, the mean benzene concentration was 1.73 ppbv, with a maximum of 8.44 ppbv. However, benzene was not significantly correlated or associated with 3-ethenylpyridine, a proposed vapor-phase environmental tobacco smoke marker (Heavner et al. 1995).

Limited monitoring values for commercial indoor air are available. A study conducted from the summer of 2003 to the winter of 2005 reported geometric means of 0.55 ppbv benzene in air of U.S. stores and 0.96 ppbv benzene in air of U.S. restaurants (Loh et al. 2006). Concentrations in restaurants were influenced by the presence of smokers and would therefore be expected to be lower today due to restrictions on smoking in public places.

Benzene may be present in indoor air by vapor intrusion from contaminated groundwater into buildings. The EPA (2016c) includes benzene 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 benzene during site risk assessments. However, there are many other background sources of benzene to indoor air pollution, and several site assessments did not find vapor intrusion to be the main source of benzene.

At a former naval air station and hazardous waste disposal site in California, the highest indoor air level of benzene was 1.18 $\mu g/m^3$ (0.369 ppbv) in an aircraft engine facility; vapor intrusion was supported as a source of benzene based on detections in groundwater and soil gas, but these concentrations were not provided in this report (ATSDR 2022c).

In Indiana, possible vapor intrusion from a plume of contaminated groundwater into commercial buildings was investigated. Benzene concentrations were <0.13–3.6 $\mu g/m^3$ (<0.04–1.1 ppbv) in the sub-slab gas and 1.7–9.8 $\mu g/m^3$ (0.53–3.1 ppbv) in soil below a recycling center, gymnastics center, and ambulance company and electrical contractor (ATSDR 2023a). Adjusted indoor air concentrations (to reflect exposure for 5 days/week, 10 hours/day) were <0.13–1.0 $\mu g/m^3$ (<0.04–0.31 ppbv). The ratio of underground benzene to indoor air benzene was 13 and not suggestive of an underground source at this site, since ratios >33 indicate underground source contribution.

Benzene concentrations in soil gas of a former cleaning products manufacturing site and current Superfund site in Texas ranged from below the detection limit to 96 μg/m^3 (30 ppbv), 0.58–7.7 μg/m^3 (0.18–2.4 ppbv) in indoor air, and 0.3 μg/m^3 (0.09 ppbv) in outdoor air onsite; concentrations in off-site soil gas were 0.1–29 μg/m^3 (0.03–9.1 ppbv) and concentrations in indoor air were 0.3–0.42 μg/m^3 (0.09–0.13 ppbv) (ATSDR 2023b). Vapor intrusion was not likely the source of benzene to indoor air at this site. Near another Superfund site that was contaminated with dry cleaner solvents in Tennessee, a restaurant adjacent to the site had benzene in soil gas up to 9.9 μg/m^3 (3.1 ppbv) and was not detected (limit of detection [LOD]: 0.48 μg/m^3; 0.15 ppbv) in the indoor air of the restaurant (ATSDR 2021).

Townhomes in Minnesota were investigated from possible vapor intrusion due to underlying groundwater contamination from a nearby NPL site (ATSDR 2023c). Benzene air levels measured in these townhomes between May and July of 2015 were 1.2–5.3 μg/m^3 (0.36–1.7 ppbv) in indoor air, 1.3–4.5 μg/m^3 (0.41–1.4 ppbv) in crawl space air, and 2.3–4.4 μg/m^3 (0.71–1.4 ppbv) in sub-slab gas. Measured benzene air levels in outdoor air near townhomes over the same time period were 0.35–0.76 μg/m^3 (0.12–0.24 ppbv). Pollution was determined likely to be from indoor sources rather than vapor intrusion (ATSDR 2023c).

Ambient air is impacted by nearby hazardous waste sites, oil spills, or other accidental releases of hazardous materials. Benzene was measured in the vicinity of the Bridgeton landfill, a solid waste landfill in Missouri, at a maximum concentration of 32.5 ppbv (ATSDR 2022b). In the 5 months following the 2010 Deepwater Horizon oil spill, mean benzene concentrations were 4.83 μg/m^3 (1.51 ppbv; range: 0.12–81.89 μg/m^3; 0.04–25.63 ppbv) in regional areas and 2.96 μg/m^3 (0.927 ppbv; range: 0.14–290 μg/m^3; 0.04-–90.8 ppbv) in coastal areas of Louisiana (IARC 2018). On February 3, 2023, a freight train carrying hazardous materials derailed in East Palestine, Ohio. Some of the cars caught fire, while others spilled their loads into an adjacent stream. In air samples collected at the train derailment site between February and August 2023, benzene was detected at a median of 0.175 ppbv (range: 0.072–3.76 ppbv) (EPA 2023c; Oladeji et al. 2023).

Ambient air is also impacted by nearby oil and gas development. Outdoor monitoring of residential areas in the Dallas/Fort Worth metro area was conducted varying distances from urban drilling and unconventional shale gas exploration and production operations (Rich and Orimoloye 2016). Five of the six counties monitored were classified as non-attainment areas during the time of sampling. The mean benzene concentration was 18.53±83.75 ppbv (range: 0.6–592 ppbv). Sampling sites were selected to avoid influence of gas station and roadway emissions, and the drilling operations were believed to be the primary source of the benzene (Rich and Orimoloye 2016).

Outdoor air samples were collected between May and December 2019 at an elementary school 1,400 feet north of a wellsite in Colorado (CDPHE 2020). Benzene in air was detected in 93% of all samples, at maximum concentrations of 2.28 ppbv for the baseline measurements (after wells had been drilled but not completed), 2.91 ppbv during hydraulic fracturing, 2.57 ppbv during milling, and 4.52 and 14.72 ppbv during two periods of hybrid-flowback production.

As part of a 2009 air toxics monitoring initiative, the outdoor air of selected schools with a mix of pollution sources were monitored for 60-day periods (EPA 2016a). In the case of Birmingham, Alabama, additional nearby communities near industry, major highways, or urban areas were also monitored. A summary of the monitoring results is presented in Table 5-10. EPA used a screening level of 9.3 ppbv (30 μg/m^3) to identify potential concern for risk of health problems from short- term exposures; the majority of the samples were below this value.

Table 5-10. Summary of Benzene in Outdoor Air (ppbv) Near Selected Schools Across the United States[a,b]

Location	Sampling period	Number of samples	Median	Range	Percent detected[c]
Birmingham, Alabama					
Lewis Elementary School	July 2009–November 2009	22	0.349	0.088–7.01	64%
North Birmingham Elementary School	July 2009–November 2009	24	0.444	0.081 –9.42	71%
Riggins School	July 2009–December 2009	27	1.610	0.131–9.55	37%
Additional community monitoring					
Collegeville	June 2011–August 2012	72	0.427	0.113 –6.86	83%
Fairmont	June 2011–August 2012	75	0.905	0.097–17.1	87%
Harriman Park	June 2011–August 2012	71	0.789	0.163–7.12	85%
North Birmingham	June 2011–August 2012	73	0.319	0.117–6.39	84%
Tarrant City, Alabama					
Tarrant Elementary School	July 2009–November 2009	22	0.363	0.078–7.26	95%

Table 5-10. Summary of Benzene in Outdoor Air (ppbv) Near Selected Schools Across the United States[a,b]

Location	Sampling period	Number of samples	Median	Range	Percent detected[c]
Los Angeles, California					
Felton Elementary School	August 2009–Mar 2010	23	0.398	0.163–1.12	87%
Soto Street Elementary School	August 2009–Mar 2010	37	0.520	0.211–0.986	41%
East Chicago, Indiana					
Abraham Lincoln Elementary School	August 2009–December 2009	13	0.149	0.094–0.319	77%
Gary, Indiana					
Jefferson Elementary School	August 2009–December 2009	16	0.186	0.101–0.304	63%
Ashland, Kentucky					
Charles Russell Elementary School	July 2009–April 2010	25	0.260	0.149–2.35	52%
Crabbe School	July 2009–April 2010	25	0.329	0.176–4.23	48%
Hatcher School	July 2009–April 2010	25	0.274	0.150–2.36	48%
Paulsboro, New Jersey					
Paulsboro High School	August 2009–February 2010	27	0.341	0.183–0.592	44%
Elizabeth, New Jersey					
Mabel Holmes Middle School	August 2009–January 2010	25	0.229	0.145–0.601	48%
New York City, New York					
Intermediate School 143	July 2009–June 2010	25	0.360	0.252–0.642	44%
Wauseon, Ohio					
Elm Street Elementary School	September 2009–November 2009	12	0.180	0.072–0.360	83%
Ironton, Ohio					
Whitwell Elementary School	July 2009–April 2010	25	0.239	0.150–0.770	40%
Warren, Ohio					
Life Skills of Trumbull County and Academy of Arts and Humanities	August 2009–February 2011	36	0.346	0.158–0.917	56%
Portland, Oregon					
Harriet Tubman Middle School	August 2009–November 2009	13	0.242	0.135–0.704	92%
Clairton, Pennsylvania					
Clairton Educational Center	September 2009–December 2009	18	0.526	0.097–2.95	56%
McKeesport, Pennsylvania					
South Allegheny Middle/High School	August 2009–April 2012	59	0.905	0.094–18.9	80%

Table 5-10. Summary of Benzene in Outdoor Air (ppbv) Near Selected Schools Across the United States[a,b]

Location	Sampling period	Number of samples	Median	Range	Percent detected[c]
McKees Rock, Pennsylvania					
Sto-Rox Elementary School and Sto-Rox Middle School	August 2009–January 2010	22	0.181	0.088–0.488	45%
North Charleston, South Carolina					
Chicora Elementary School	July 2009–March 2010	40	0.314	0.146–0.679	25%
Dallas, Texas					
NW Harllee Elementary School	September 2009–December 2009	15	0.194	0.124–0.711	67%
Deer Park, Texas					
San Jacinto Elementary School and Deer Park Junior High School	September 2009–May 2012	56	0.534	0.135–3.79	79%
Houston, Texas					
Young Scholars Academy	September 2009–December 2009	16	0.267	0.226–0.457	63%
Seattle, Washington					
Concord Elementary School	August 2009–November 2009	19	0.284	0.154–0.545	68%
Follansbee, West Virginia					
Follansbee Middle School and Jefferson Primary School	August 2009–August 2012	66	0.826	0.129–6.32	73%
Durango, Colorado (tribal area)					
Sunnyside Elementary School	August 2009–November 2009	15	0.129	0.072–0.256	93%
Lapwai, Idaho (tribal area)					
Lapwai High School	September 2009–December 2009	16	0.152	0.078–0.326	81%

[a] Values were originally reported in micrograms per cubic meter (μg/m^3) and converted to ppbv.
[b] 24-hour sampling period.
[c] Minimum limit of detection: 0.0006 ppbv.

Source: EPA 2016a

5.5.2 Water

The EPA maintains a Water Quality Portal (WQP) database which aggregates air monitoring data from the National Water Information System (NWIS) and STORage and RETrieval (STORET) system. A summary of the data for ambient surface and groundwater from recent years are reported in Table 5-11

(WQP 2023). Benzene was detected at higher concentrations and at greater frequency in groundwater than in surface water.

Table 5-11. Summary of Benzene Concentrations (ppb) in Surface and Groundwater Across the United States

Year	Average	Maximum	Number of samples	Percent detected
Surface water				
2019	0.420	7.8	1,279	9.1%
2020	0.473	1.23	824	7.4%
2021	0.012	0.014	554	0.7%
2022	0.184	0.5	448	0.7%
2023[a]	0.038	0.04	86	2.3%
Groundwater				
2019	5.00	300	1,812	4.1%
2020	159	4,580	1,313	4.9%
2021	1,407	27,000	1,729	15.7%
2022	1,003	24,500	2,774	16.1%
2023[a]	765	16,800	1,126	15.6%

[a] As of December 7, 2023.

Source: WQP 2023

The U.S. Geological Survey (USGS) conducted a national assessment of 55 VOCs in well water samples collected from 2,401 domestic wells around the country during 1985–2002. Benzene was detected in 37 of 1,208 well samples, or 3.1% of the samples, at concentrations mostly <1 µg/L (Rowe et al. 2007). During another assessment of principal aquifers in the United States conducted from 1991 to 2010, benzene was detected in 2.05% of areas sampled that are used for drinking water, 1.47% of shallow groundwater beneath agricultural land, and 5.80% of shallow groundwater beneath urban land (USGS 2014). The median detected concentration was 0.035 ppb (range: 0.01–290 ppb).

In a USGS-conducted groundwater quality monitoring campaign, benzene was detected in 5 of the 336 samples collected between 2013 and 2016 from wells across the United States: 0.014 ppb in one well in South Carolina, 0.021 ppb in one well in Kansas, 0.052 in one well in Arkansas, 0.244 ppb in one well in Oklahoma, and 1.35 ppb in one well in Tennessee (USGS 2020).

Benzene is not a common contaminant of municipal water. EPA conducted their third 6-year compliance monitoring assessment of public water supplies between 2006 and 2011 (EPA 2016b). Benzene was detected in 0.31% of the 372,470 public water supply samples analyzed, with a median value of 0.8 ppb (range: 0.02–230 ppb).

Drinking water can be impacted indirectly by forest fires. Following the 2017 Tubbs Fire in California, benzene was detected at an average of 11.0 ppb in municipal water of Santa Rosa city (maximum: 40,000 ppb 6 months after the fire) during sampling 1–20 months after the fire (Proctor et al. 2020a, 2020b). Eight months following the 2018 Camp Fire in California, benzene was detected at a maximum of 923 ppb in municipal waters. The study authors suggested that thermal degradation of plastic pipes may have released benzene into the water system. Another study following the 2018 Camp Fire reported a mean of 18.97±75.26 ppb benzene in municipal water sampled in Paradise, California, from December 2018 to May 2020 (Solomon et al. 2021).

Proximity to hazardous waste sites, oil spills, or other accidental releases of hazardous materials may result in increased benzene in surface water, groundwater, and drinking water derived from these local sources, depending on the site uses. During a site assessment of a former army air and ground base in Port Heiden, Alaska, benzene was reported in groundwater near a wastewater drainage pond and near a former fuel pipeline, at a maximum concentration of 9.4 ppb (ATSDR 2019a). The groundwater near the pond site was not used for drinking water but some wells near the fuel pipe previously were. Other samples collected near the fuel pipeline corridor ranged from not detected to <0.15 ppb (reporting limit) benzene in groundwater and a maximum of 0.98 ppb benzene in surface water (ATSDR 2019a). Benzene was not detected in public drinking water on St. Lawrence Island, Alaska, after investigation based on concerns of fuel contaminants resulting from the former military surveillance and communications station (ATSDR 2020).

Benzene was detected at a maximum of 7.65 ppb in drinking water sourced from groundwater wells and springs impacted by natural gas drilling in Pennsylvania after unauthorized release of chemicals in order to recover a lost drill bit (ATSDR 2019b). At a former truck terminal and tank-trailer cleaning facility NPL site in New Jersey, which disposed of wastewater in an unlined lagoon and underground tanks onsite, benzene was detected at 92 ppb in monitoring wells near the former lagoon area and in one surface water sample (n=17) at 1.80 ppb at off-site locations along the Grand Sprute Run tributary (ATSDR 2023d).

Benzene was detected at 0.5 ppb in only one groundwater sample used for drinking water near the Dorado NPL site in Puerto Rico (ATSDR 2023e). It was not detected in the aqueduct used for drinking water distribution. Benzene was not detected in groundwater samples (n=82) collected between 2018 and 2020 at the Palermo Wellfield Superfund site (WQP 2023), nor was it detected in surface water samples (reporting limits: 0.27–100 ppb) collected in February 2023 at the East Palestine, Ohio train derailment site (EPA 2023c).

Historically, benzene has been detected in water in the vicinity of industrial facilities using or producing benzene from <1 ppb to a high of 179 ppb (found in plant effluent). In general, benzene in plant effluents quickly dispersed in rivers or streams to levels of 1–≤2 ppb (EPA 1979). Benzene concentrations monitoring wells within aquifers near fuel spills at gasoline service stations were 1,200–19,000 ppb (Salanitro 1993). A monitoring well in the vicinity of a bulk storage facility had a maximum benzene level of 45,000 ppb (Salanitro 1993). In northern Virginia, approximately 200,000 gallons of liquid hydrocarbons were released from a fuel-storage terminal into the underlying soil. A dichloromethane extract of groundwater from a monitoring well in the same area gave a benzene concentration of 52.1 ppm (Mushrush et al. 1994). Benzene has been detected at concentrations of 16–110 ppb in landfill leachate from a landfill that accepted both municipal and industrial wastes (Cline and Viste 1985).

Marine water contains high benzene concentrations when an oceanic oil spill occurs. In water column samples obtained from the Deepwater Horizon oil spill plume, benzene concentrations were 0.4–21.7 ppb at ~1,100 m depths; however, benzene was not detected at depths less than 1,000 m (Reddy et al. 2012).

5.5.3 Sediment and Soil

Limited ambient soil and sediment monitoring data are available. Benzene was not detected in sediment (n=52) and soil (n=14) samples reported between 2019 and 2023 in the EPA Water Quality Portal database (WQP 2023).

Some recent data are available regarding the presence of benzene at hazardous waste sites. Benzene was reported in soil near the marine terminal area of a former army air and ground base in Port Heiden, Alaska, at a maximum of 0.026 ppb (ATSDR 2019a). At an NPL site of a former truck terminal with tank-trailer cleaning in New Jersey, which disposed of wastewater in an unlined lagoon and underground tanks onsite, benzene concentrations were 0.03–39 ppm in sediment samples at off-site locations along the Grand Sprute Run tributary (ATSDR 2023d). Benzene was not detected in sediment samples (n=5) collected in 2021 at the Palermo Wellfield Superfund Site (WQP 2023). Benzene was not included in soil analysis at the East Palestine, Ohio, train derailment site (EPA 2023c).

Historically, benzene levels of <2–191 ppb in soil were recorded in the vicinity of industrial facilities using or producing benzene (EPA 1979). In northern Virginia, following the release of approximately 200,000 gallons of liquid hydrocarbons from a fuel-storage terminal into the underlying soil, benzene was detected in soil at a concentration of 1,500 ppm at a depth of 10 feet, about 1,000 feet from the storage terminal (Mushrush et al. 1994).

5.5.4 Other Media

Benzene has been detected in a variety of food and beverages. Benzene may form in food and beverages from benzoate salts (e.g., sodium benzoate and potassium benzoate), which are added as a preservative or are naturally present such as in some fruits, and ascorbic acid (Vitamin C), which may also be added or be naturally present (Meadows 2006; Medeiros Vinci et al. 2012). Conditions such as low sugar, low acidity, and storage conditions under strong light and higher temperatures promote benzene formation; changing the product formulation and storage conditions successfully reduces the likelihood of benzene contamination (Salviano Dos Santos et al. 2015). Foods may also absorb benzene during processes such as smoking (Medeiros Vinci et al. 2012) or from VOCs released during cooking with oil (Medeiros Vinci et al. 2012; Pellizzari et al. 1995).

A summary of the available data for benzene in food and beverages is reported in Table 5-12. The most recent large-scale study in the United States was an FDA-sponsored 5-year study of table-ready foods from 1996 to 2000; foods tested included both foods that were purchased uncooked (and cooked prior to testing) as well as ready-to-eat food purchases (Fleming-Jones and Smith 2003). Benzene was detected in all sampled foods except for American cheese and vanilla ice cream. More recent data were available from a sampling of Belgian supermarkets in 2010 (Medeiros Vinci et al. 2012). Selected products not otherwise covered by Fleming-Jones and Smith (2003), and products with high instances of detection are reported in Table 5-12; the products with the highest detected benzene were smoked, canned, and raw fatty fish, while non-fatty fish and raw meat had lower levels.

Table 5-12. Summary of Benzene in Food and Beverages

Location (sampling period)	Product	Mean (ppb)	Range (ppb)	Notes	Source
United States, 1996–2000	Cheddar cheese		20–47	Detected in 2 samples	Fleming-Jones and Smith 2003[a]
	American cheese		–	Not detected in an unknown number of samples	
	Mixed nuts		1–6	Detected in 3 samples	
	Ground beef		9–190	Detected in 12 samples	
	Pork bacon		2–17	Detected in 6 samples	
	Banana, raw		11–132	Detected in 13 samples	
	Frankfurters, beef		2–11	Detected in 4 samples	
	Cream cheese		1–17	Detected in 3 samples	
	Chocolate cake icing		2–23	Detected in 2 samples	
	Tuna canned in oil		4–13	Detected in 7 samples	
	Fruit flavored cereal		2–21	Detected in 5 samples	
	Eggs scrambled		2–40	Detected in 4 samples	
	Peanut butter		2–25	Detected in 5 samples	
	Avocado, raw		3–30	Detected in 10 samples	
	Popcorn, popped in oil		4–22	Detected in 3 samples	
	Blueberry muffin		3–8	Detected in 3 samples	
	Strawberries, raw		1	Detected in 1 sample	
	Cola carbonated		1–138	Detected in 3 samples	
	Orange, raw		11–15	Detected in 2 samples	
	Coleslaw with dressing		11–102	Detected in 14 samples	
	Sweet roll Danish		3	Detected in 1 sample	
	Potato chips		2–7	Detected in 2 samples	
	Fruit flavored sherbet		3–61	Detected in 3 samples	
	Quarter pound hamburger cooked		4–47	Detected in 11 samples	
	Margarine		7	Detected in 1 sample	
	Sandwich cookies		1–39	Detected in 3 samples	
	Butter		4-22	Detected in 6 samples	
	Chocolate chip cookies		1–8	Detected in 2 samples	
	Sour cream		3–15	Detected in 2 samples	
	Apple pie fresh/frozen		2–11	Detected in 4 samples	
	Chicken nuggets fast food		2–5	Detected in 4 samples	
	Graham crackers		1–9	Detected in 2 samples	
	French fries, fast food		2–56	Detected in 3 samples	
	Cheeseburger quarter pound		5–44	Detected in 8 samples	
	Cheese pizza		1–2	Detected in 2 samples	
	Bologna		2–44	Detected in 4 samples	

Table 5-12. Summary of Benzene in Food and Beverages

Location (sampling period)	Product	Mean (ppb)	Range (ppb)	Notes	Source
	Cheese and pepperoni pizza		8–30	Detected in 4 samples	
	Olive/safflower oil		1–46	Detected in 6 samples	
	Sugar cookies		8–30	Detected in 3 samples	
	Cake doughnuts with icing		3	Detected in 2 samples	
	Vanilla ice cream		–	Not detected in an unknown number of samples	
	Popsicle		1–10	Detected in 4 samples	
Belgium, 2010	Ready for consumption beverages	1.30	NS–11.52	Detected in 14/28 samples	Medeiros Vinci et al. 2012
	Smoothies and juices	–	0.48	Detected in 1/9 samples	
	Coffee, tea, and infusions	0.21	NS–1.33	Detected in 2/12 samples	
	Water (carbonated)	–	–	Not detected in 3 samples	
	Sauces	1.18	NS-–6.35	Detected in 11/14 samples	
	Dips and tapenades	1.44	NS–6.48	Detected in 15/17 samples	
	Non-refined oils	2.19	NS–5.04	Detected in 11/12 samples	
	Refined oils	0.74	NS–2.55	Detected in 11/11 samples	
	Fresh fruit	–	–	Not detected in 10 samples	
	Fresh vegetables	–	–	Not detected in 10 samples	
	Canned fruits and vegetables	–	–	Not detected in 11 samples	
	Smoked fish	18.90	NS–76.21	Detected in 10/10 samples	
	Canned fish	7.40	NS–20.81	Detected in 15/16 samples	
	Raw fatty fish	3.15	NS–9.44	Detected in 4/4 samples	
	Raw non-fatty fish	0.52	NS–2.00	Detected in 2/6 samples	
	Processed (dried/cured/fermented) meat	1.77	NS-–6.95	Detected in 6/7 samples	
	Raw meat	–	1.25	Detected in 1/4 samples	
	Fresh eggs	–	–	Not detected in 9 samples	
	Chilled ready to eat meals	1.11	NS-3.68	Detected in 11/14 samples	
	Ready to eat salads	2.79	NS-25.46	Detected in 62/68 samples	

Table 5-12. Summary of Benzene in Food and Beverages

Location (sampling period)	Product	Mean (ppb)	Range (ppb)	Notes	Source
Maryland, Virginia, and Michigan, November 2005–May 2007	Beverages containing benzoate only	1.75	1.1–3.5	Quantifiable in 4/61 samples; detected (>0.2 ppb) but not quantifiable (<1 ppb) in 26/61 samples	FDA 2015
	Beverages containing benzoate and ascorbic acid or erythorbic acid	12.3	1.1–87.9	Quantifiable in 55/102 samples; detected (>0.2 ppb) but not quantifiable (<1 ppb) in 28/102 samples	
	Beverages containing ascorbic acid or erythorbic acid	2.70	1.3–4.8	Quantifiable in 7/24 samples; detected (>0.2 ppb) but not quantifiable (<1 ppb) in 9/24 samples	
	Other beverages (e.g., juice cocktails, may contain natural benzoate)	4.96	1.6-10.7	Quantifiable in 6/12 samples; detected (>0.2 ppb) but not quantifiable (<1 ppb) in 4/12 samples	

[a] Only foods containing over 100 ppb at least one VOC were reported by study authors.

NS = not stated; VOC = volatile organic compound

The FDA analyzed several soft drink and juice beverages between 2005 to 2007 due to concerns of benzene formation from preservatives and additives (FDA 2015; Meadows 2006). Twenty-three samples contained benzene at concentrations >5 ppb (EPA's standard for drinking water); FDA stated that they would follow up with manufacturers about product reformulation (FDA 2015).

Between 2021 and 2023, 17 products were recalled by the FDA for the presence of benzene (FDA 2023b). These products included hand sanitizers and spray products such as an antifungal spray powder, anesthetics, antiperspirants, dry shampoo, and sunscreen. Benzene may be present in these consumer products as a residual contaminant from petroleum-based feedstocks sometimes used during product manufacturing (Kumar Pal et al. 2022). For spray consumer products, contamination is likely due to isobutane spray propellants used in these products (FDA 2023c). Contamination in hand sanitizers became of greater concern during the Covid-19 epidemic, when use and demand increased dramatically. Commercially available hand sanitizers purchased online or available in public places (e.g., restaurants and hospitals) in New York State from April 2021 to July 2021 had a median of 0.081 ppb benzene (range: 0.081–22,300 ppb) detected (Kumar Pal et al. 2022).

Benzene is an incomplete combustion product of organic material, which may be a significant source of pollution to indoor environments. In one study analyzing emission rates from burning wood in a fireplace, pine emitted 383 mg benzene per kg of wood burned (Schauer et al. 2001). Cooking stoves are another significant emitter of benzene. Median emission rates ranged from 0.04 μg benzene emitted per minute for induction heat stovetops on high to 3.89 μg benzene emitted per minute for propane ovens set to 350°F (Kashtan et al. 2023). Furthermore, there were benzene emissions from electric coils and ovens and radiant stoves due to burning of residual organic material at high temperatures. In 9 of the 33 cases, a single gas burner on high or gas oven set to 350°F raised kitchen air benzene concentrations above those expected from second-hand smoke (>0.78 ppbv).

Cigarette smoke remains an important source of human exposure to benzene. The amount of benzene measured in mainstream smoke was 5.9–73 μg/cigarette (Brunnemann et al. 1990). Larger amounts of benzene (345–653 μg/cigarette) were found in sidestream smoke (Brunnemann et al. 1990). Benzene has been found in vapor from cigarette smoke at concentrations of 3.2–61.2 μg/cigarette depending on the brand of cigarette. The amount of tar in the cigarette was not directly related to the benzene concentration (Darrall et al. 1998). Benzene has been detected in the emissions of electronic cigarettes at 0.5–2.6 ppb (one puff per minute) and 0.7–6.6 ppb (two puffs per minute) (Lee et al. 2017).

Benzene monitoring data in biota were not located. Based on experimental results, benzene is not expected to accumulate in animals or plants to a significant degree (see Section 5.4.1).

5.6 GENERAL POPULATION EXPOSURE

The general population is environmentally exposed to benzene primarily through ambient air, and in rarer instances, through water and drinking water. Benzene has been detected at low levels in food as a result of cooking or formation from added preservatives. The most significant general population exposures are in indoor air from sources such as evaporative emissions from cars in attached garages, or from cooking on gas stoves, from smoking, and from automobile related activities such as pumping gasoline.

The CDC conducts continuous monitoring of the non-institutionalized, civilian U.S. population during 2-year study periods under the NHANES program, beginning in 1999. Whole-blood benzene results from the most recent survey period are summarized in Table 5-13. An additional survey comparing the blood benzene levels of smokers versus nonsmokers was conducted, the most recent results are summarized in Table 5-14. In these recent results, median whole-blood levels of benzene in the general population or in nonsmokers could not be determined because values were below the level of detection (0.024 ng/mL). The median whole-blood benzene level in smokers was 0.178 ng/mL.

Table 5-13. Benzene in Whole-Blood Samples (ng/mL) of the U.S. Population (2017–2018)

Population group	Geometric mean	50th percentile (95% CI)	75th percentile (95% CI)	90th percentile (95% CI)	95th percentile (95% CI)	Sample size
Total population	NA	<LOD	0.034 (0.025–0.046)	0.138 (0.112–0.175)	0.263 (0.207–0.319)	2,840
Age 12–19 years	NA	<LOD	<LOD	0.038 (0.027–0.046)	0.048 (0.040–0.063)	472
Age 20+ years	NA	<LOD	0.038 (0.029–0.052)	0.155 (0.121–0.199)	0.282 (0.221–0.366)	2,368
Males	NA	<LOD	0.039 (0.028–0.059)	0.161 (0.118–0.204)	0.279 (0.210–0.335)	1,380
Females	NA	<LOD	0.029 (<LOD–0.042)	0.117 (0.079–0.150)	0.245 (0.162–0.335)	1,460
Mexican American	NA	<LOD	0.027 (<LOD–0.044)	0.054 (0.042–0.069)	0.079 (0.061–0.143)	412
Non-Hispanic Black	NA	<LOD	0.075 (0.054–0.102)	0.212 (0.179–0.263)	0.334 (0.273–0.399)	636
Non-Hispanic White	NA	<LOD	0.033 (<LOD–0.052)	0.137 (0.086–0.190)	0.279 (0.180–0.366)	972
All Hispanic	NA	<LOD	0.026 (<LOD–0.038)	0.058 (0.044–0.079)	0.111 (0.069–0.146)	672
Non-Hispanic Asian	NA	<LOD	<LOD	0.032 (0.025–0.051)	0.059 (0.041–0.083)	396

CI = confidence interval; LOD = limit of detection (0.024 ng/mL); NA = not available (proportion of results below limit of detection was too high to provide a valid result)

Source: CDC 2022a

Table 5-14. Benzene in Whole Blood Samples (ng/mL) of the U.S. Smoking and Nonsmoking Populations (2015–2016)

Population group	Geometric mean (95% CI)	50th percentile (95% CI)	75th percentile (95% CI)	90th percentile (95% CI)	95th percentile (95% CI)	Sample size
Total smoking population	0.153 (0.135–0.174)	0.178 (0.163–0.198)	0.308 (0.280–0.338)	0.455 (0.395–0.535)	0.642 (0.510–0.826)	824
Age 18–49 years	0.121 (0.096–0.153)	0.144 (0.111–0.178)	0.261 (0.213–0.336)	0.407 (0.347–0.503)	0.602 (0.425–0.826)	456
Age 50+ years	0.224 (0.196–0.256)	0.224 (0.198–0.261)	0.372 (0.303–0.395)	0.541 (0.395–0.796)	0.796 (0.503–1.37)	368

Table 5-14. Benzene in Whole Blood Samples (ng/mL) of the U.S. Smoking and Nonsmoking Populations (2015–2016)

Population group	Geometric mean (95% CI)	50th percentile (95% CI)	75th percentile (95% CI)	90th percentile (95% CI)	95th percentile (95% CI)	Sample size
Males	0.142 (0.123–0.164)	0.169 (0.145–0.185)	0.287 (0.243–0.346)	0.477 (0.395–0.625)	0.741 (0.602–1.01)	494
Females	0.168 (0.145–0.194)	0.188 (0.169–0.213)	0.330 (0.280–0.354)	0.453 (0.379–0.523)	0.541 (0.415–0.796)	330
Total nonsmoking population	NA	<LOD	0.028 (<LOD–0.036)	0.050 (0.042–0.058)	0.067 (0.058–0.080)	2,045
Age 18-49 years	NA	<LOD	0.029 (<LOD–0.039)	0.053 (0.043–0.067)	0.078 (0.057–0.130)	1,031
Age 50+ years	NA	<LOD	0.027 (<LOD–0.036)	0.047 (0.039–0.054)	0.062 (0.051–-0.071)	1,014
Males	NA	<LOD	0.029 (<LOD–0.038)	0.051 (0.041–0.057)	0.070 (0.057–0.086)	924
Females	NA	<LOD	0.026 (<LOD–0.035)	0.050 (0.039–0.062)	0.065 (0.054–0.089)	1,121

CI = confidence interval; LOD = Limit of detection (0.024 ng/mL); NA = not available (proportion of results below limit of detection was too high to provide a valid result)

Source: CDC 2022a

Historical median whole-blood benzene levels (NHANES III, 1988–1994) were 0.061 ng/mL in the general population and 0.047 ng/mL in nonsmokers only (Ashley et al. 1994; Lemire et al. 2004). For previous NHANES data, significantly higher median blood concentrations were seen in individuals who had pumped gasoline into a car or other motor vehicle, and for those who inhaled diesel exhaust, in the 3 days leading up to sampling compared to those who hadn't (IARC 2018).

Because blood benzene is sensitive to recent exposures, *trans,trans*-muconic acid and PhMA, urinary metabolites of benzene, are used as biomarkers of environmental and occupational exposure (IARC 2018). The CDC conducts urinary monitoring as part of the NHANES program, and the results of the most recent survey years are reported in Table 5-15.

Table 5-15. Urinary Metabolites of Benzene (ng/mL) in the U.S. Population (2017–2018)

Population group (sex, age)	Geometric mean	95% CI	Samples below the MDL	Sample size
trans,trans-Muconic acid				
Total population	60.8	58.8–62.8	177	4,734
Age 3–5 years	66.4	58.9–74.8	16	543
Age 6–11 years	89.6	80.2–100	18	568
Age 12–19 years	57.4	51.9–63.5	19	641
Age 20+ years	56.2	53.9–58.7	57	2,982
Male	71.2	67.8–74.8	53	2,350
Female	52.0	49.4–54.7	124	2,384
Mexican American	60.3	54.7–66.5	30	665
Non-Hispanic Black	66.4	61.8–71.3	24	1,058
Non-Hispanic White	71.2	67.0–75.7	45	1,667
Non-Hispanic Asian	33.0	29.9-36.4	50	554
PhMA				
Total population	0.194	0.189–0.200	1,451	4,734
Age 3–5 years	0.148	0.141–0.156	105	543
Age 6–11 years	0.152	0.145–0.159	116	568
Age 12–19 years	0.151	0.144–0.159	117	641
Age 20+ years	0.226	0.217–0.235	522	2,982
Male	0.212	0.204–0.221	642	2,350
Female	0.178	0.172–0.185	809	2,384
Mexican American	0.159	0.151–0.167	238	665
Non-Hispanic Black	0.233	0.219–0.249	255	1,058
Non-Hispanic White	0.206	0.196–0.217	498	1,667
Non-Hispanic Asian	0.141	0.134–0.148	222	554

CI = confidence interval; MDL = method detection limit (9.81 ng/mL for *trans,trans*-muconic acid and 0.150 ng/mL for PhMA); PhMA = *S*-phenylmercapturic acid

Source: CDC 2022b

Between 1999 and 2000, as part of the NHANES, the CDC conducted personal air monitoring via passive exposure monitors (or badges) for 48–72-hour periods, followed by an exposure factors questionnaire (CDC 2005). A summary of the results is presented in Table 5-16. Significantly higher benzene exposures were observed for participants who had homes with attached garages versus participants who did not have an attached garage; participants who had no windows open in the home versus participants who had any open windows; and participants who were smokers versus nonsmoking participants (Symanski et al. 2009). More recent general population personal air monitoring data have been reported from the Detroit Exposure and Aerosol Research Study, which measured ambient breathing zone concentrations of benzene for from 2004 to 2007 in Detroit, Michigan (George et al. 2011). The mean average daily personal benzene air concentrations were 2.7–7.7 μg/m^3 (858 samples).

Table 5-16. Benzene in Personal Air Monitoring (μg/m^3) of the U.S. Adult Population (1999–2000)

Population group (sex, age)	Geometric mean	95% CI	Samples below the MDL	Sample size
Total population	3.25	3.03–3.49	222	669
Age 20–29 years	3.10	2.70–3.57	75	185
Age 30–39 years	3.35	2.93–3.83	58	172
Age 40–49 years	3.44	2.98–3.96	42	158
Age 50+ years	3.14	2.65–3.79	47	132
Male	3.74	3.35–4.17	84	295
Female	2.90	2.64–3.18	138	352
Mexican American	4.30	3.77–4.91	34	185
Other Hispanic	2.88	2.48–3.35	55	126
Non-Hispanic Black	2.84	2.55–3.18	108	276
Non-Hispanic White	3.03	2.34–3.91	21	45

CI = confidence interval; MDL = method detection limit (not reported)

Source: CDC 2005

Nationally, a decline in ambient air benzene of 66% has been reported between 1996 and 2009 (EPA 2014). Average intake from outdoor air is estimated to be 0.16 μg/kg body weight/day based on an average concentration of 0.562 μg/m^3 (0.176 ppbv) (EPA 2023b) and an inhalation rate of 20 m^3/day for a 70-kg person.

Benzene is not a common pollutant of municipal water, and drinking water is not expected to be an important route of exposure for benzene (Wallace 1989a). A median value of 0.8 μg/L has been reported based on benzene detections in 0.31% of the 372,470 public water supply samples analyzed (EPA 2016b). Based on this value, a water intake of 0.02 μg/kg body weight/day is estimated for these populations based on a water consumption rate of 2 L/day for a 70-kg person.

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 are 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. While benzene is not commonly detected in municipal water, it is expected to be readily volatile from water and exposure during bathing may occur from trace amounts of benzene present for some populations. Using a median treated water concentration of 0.8 μg/L and a representative outdoor air level of 0.562 μg/m^3 (0.176 ppbv), Reasonable Maximum Exposure (RME) levels were calculated for different exposure groups and are reported in Table 5-17.

Table 5-17. Estimated RME Daily Inhalation Dose and Administered Dermal Dose for the Target Person

Exposure group	Inhalation dose (μg/kg/day)	Inhalation dose (μg/m^3)	Dermal dose (μg/kg/day)
Birth–<1 year	1.2	1.2	0.0050
1–<2 years	1.2	1.2	0.0047
2–<6 years	0.85	1.2	0.0040
6–<11 years	0.50	1.2	0.0032
11–<16 years	0.34	1.2	0.0026
16–<21 years	0.27	1.2	0.0024
Adult	0.24	1.2	0.0024
Pregnant and breastfeeding women	0.35	1.2	0.0024

Source: ATSDR 2023f

Benzene may be present at trace amounts in some food and dietary intake is not expected to be a significant route of exposure for the general population (Wallace 1989a). A relatively recent study detected benzene in 58% of the 455 food samples collected from the Belgian market. The highest concentrations of benzene were found in processed foodstuffs, including fish (smoked or canned) with a reported maximum concentration of 76.21 μg/kg; raw meat, fish, eggs, and other unprocessed foods had no or lower concentrations of benzene (Medeiros Vinci et al. 2012). Using this study, an estimated mean benzene intake for all foods in the Belgian market was reported as 0.020 μg/kg body weight/day. Another dietary estimate of 0.003–0.050 μg/kg body weight/day has been reported (Salviano Dos Santos et al. 2015). The total concentration of benzene on exposed food crops consumed by humans was estimated to be 0.587 μg/kg (Hattemer-Frey et al. 1990).

Inhalation of indoor air is a major route of exposure to benzene. Indoor air benzene concentrations are typically greater than outdoor air (George et al. 2011; Kinney et al. 2002; Weisel et al. 2008). Sources include evaporative emissions of gasoline in cars, particularly in attached garages, heating sources, and cooking. Depending on airflow from garage to living areas, mean indoor benzene concentrations in houses with a garage were 2–5 times higher than outdoor levels in most homes (Thomas et al. 1993). Benzene levels in four garages during different times in a day were 0.94–61.3 ppbv (3.0–196 μg/m^3). The higher concentrations of benzene in these garages were not only from vehicular activity, but also in varying proportions from stored gasoline, paints, and benzene-containing consumer products (Thomas et al. 1993). Inhalation exposure to off-gassing from benzene-containing products and evaporative emissions from automobiles in attached garages has been estimated to be 150 μg/day (Wallace 1989a).

Benzene has been detected in residences where fuel oil was used for heating and in residences with active wood burning fireplaces (Maine DEP 2014; NYSDOH 2005; Schauer et al. 2001). Benzene has been found to be a major component of the emissions from wood burning, especially from efficient flame combustion, and constituted roughly 10–20% by weight of total non-methane hydrocarbons (Barrefors and Petersson 1995). Emissions of 383 mg benzene per kg of wood burned in a fireplace have been reported (Schauer et al. 2001). It should be noted, however, that chimney emissions result in much lower human exposure than equally large emissions at the ground level. Emissions rates of benzene during cooking on stoves or with ovens have been reported at medians of 0.04 μg benzene emitted per minute for induction hobs on high to 3.89 μg benzene emitted per minute for propane ovens set to 350°F (Kashtan et al. 2023). Even electric appliances can emit benzene from burning off of residual organic material. In some cases, a single gas burner on high or gas oven set to 350°F raised kitchen air benzene concentrations above those expected from second-hand smoke (>0.78 ppbv; >2.5 μg/m^3).

Benzene rapidly volatilizes and there is potential for vapor intrusion as a source of indoor air pollution. However, the majority of recent ATSDR site assessments did not find vapor intrusion as the source of benzene to indoor air (ATSDR 2021, 2023b, 2023c). ATSDR extracted environmental data from 135 ATSDR reports evaluating the vapor intrusion pathway at 121 sites published between 1994 and 2009 (Burk and Zarus 2013). Benzene was detected in the indoor air of 28 sites; only 2 of these sites were declared a public health hazard. EPA's compilation of background indoor air concentrations reported background medians ranging from below the reporting level (0.05–1.6 μg/m^3; 0.02–0.50 ppbv) to 4.7 μg/m^3 (1.5 ppbv), 90th percentiles of 5.2–15 μg/m^3 (1.6–4.7 ppbv), and maximum values of 21–460 μg/m^3 (6.6–140 ppbv), based on 2,615 U.S. resident samples collected between 1990 and 2005 (EPA 2011). Benzene levels in indoor air from the recent site assessments were 0.3–7.7 μg/m^3 (0.1–2.4 ppbv), generally within the median background ranges (ATSDR 2023b).

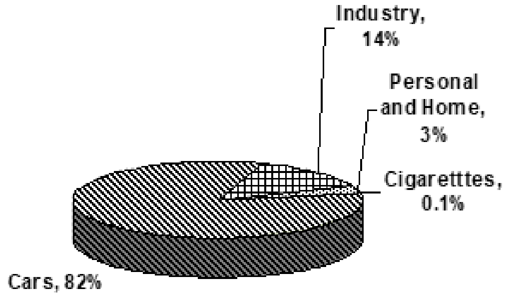
The Total Exposure Assessment Monitoring (TEAM) studies, carried out by the EPA between 1980 and 1990, suggested that for many chemicals, including benzene, the most important sources of pollution are small and close to the person, and that exposures are not clearly correlated with emissions. For example, the TEAM study findings indicated that nearly 85% of atmospheric benzene in outdoor air is produced by cars burning petroleum products and the remaining 15% is produced by industry (these estimates are expected to differ today as gasoline emissions have decreased dramatically in recent years). Despite the fact that petroleum products contributed to the majority of benzene in the atmosphere, the study found that half of the total national personal exposure to benzene came from cigarette smoke (Wallace 1995). In the United States in 2021, approximately 46 million (18.7%) adults used any tobacco product (CDC 2023).

Even passive exposure to cigarette smoke is responsible for more benzene exposure (about 5% of the total) than the emissions from the entire industrial capacity of the United States (about 3% of the total) (Wallace 1995). A breakdown of the emissions and exposure sources for benzene that was derived from the Los Angeles TEAM study data (Wallace et al. 1991) is provided in Figure 5-3. The reason that a relatively small source of emissions can have such a large effect on exposure is the efficiency of delivery. Wallace (1995) reports that one cigarette delivers an average of 55 μg of benzene with nearly 100% efficiency to the smoker. Benzene from industrial sources is dissipated into the atmosphere.

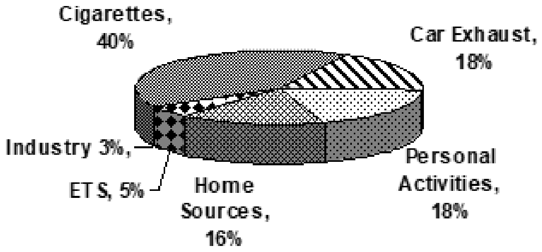
Smokers (n=200) in the TEAM study had a mean breath concentration of 15 $\mu g/m^3$ (4.7 ppbv), almost 10 times the level of 1.5–2 $\mu g/m^3$ (0.47–0.63 ppbv) observed in >300 nonsmokers (Wallace 1989b). Smokers also had about 6–10 times as much benzene in their blood as nonsmokers (Wallace 1995). In another study, benzene concentrations were compared in the breath of smokers and nonsmokers and in ambient air in both an urban area of San Francisco and in a more remote area of Stinson Beach, California (Wester et al. 1986). In the urban area, benzene in smokers' breath (6.8±3.0 ppbv) was greater than in nonsmokers' breath (2.5±0.8 ppbv) and smokers' ambient air (3.3±0.8 ppbv). In the remote area, the same pattern was observed. This suggests that smoking represented an additional source of benzene above that of outdoor ambient air (Wester et al. 1986). In 10 of 11 homes inhabited by tobacco smokers, mean indoor and personal benzene concentrations were 2–5 times higher than outdoor levels (Thomas et al. 1993).

Figure 5-3. Benzene Emissions and Exposures

Benzene Emissions



Benzene Exposures



Note: A comparison of benzene emission sources (home sources include paints and petroleum products; personal activities include driving and use of consumer products that contain benzene).

Source: Wallace et al. 1991

Assuming that the average sales-weighted tar and nicotine cigarette yields 57 μg benzene in mainstream smoke, Wallace (1989a) estimated that the average smoker (32 cigarettes/day) takes in about 1.8 mg benzene/day from smoking. This is nearly 10 times the average daily intake of nonsmokers (Wallace 1989a). On the assumption that intake of benzene from each cigarette is 30 μg, Fishbein (1992) calculated that a smoker who consumes two packs of cigarettes per day will have an additional daily intake of 1,200 μg.

A British study of rural and urban environments suggested that benzene exposure is greatly affected by proximity to smokers (Duarte-Davidson et al. 2001). Air concentrations of benzene at an urban center in South Hampton averaged about 2.5 ppbv (8.0 $\mu g/m^3$), while in a rural location in Hartwell, the average amount of benzene in the air was 0.41 ppbv (1.3 μg/m^3). Air at a smoky pub was found to contain 22 ppbv (70 μg/m^3) of benzene. Comparing the daily doses of rural nonsmokers, urban nonsmokers, urban passive smokers, and urban smokers, very little difference between the rural nonsmokers' 24 ppbv (77 μg/m^3) daily dose and the urban nonsmokers' 30 ppbv (96 μg/m^3) daily dose was found. Passive urban smokers, on average, have a daily benzene exposure dose of 38 ppbv (120 μg/m^3) of benzene while smokers have a daily exposure dose to benzene of 163 ppbv (521 μg/m^3). On average, nonsmokers in urban and rural environments have estimated benzene intakes of 1.15 and 1.5 μg/kg body weight/day.

Women tend to intake more of benzene per kg body weight than men. Passive smokers' estimated daily intake averages are 2.10 and 1.74 μg/kg body weight/day for women and men, respectively. Urban women and men smokers' estimated intakes are estimated at 9.00 and 7.46 μg/kg body weight/day, respectively; this is equivalent to an atmospheric concentration of 8.2 ppb (Duarte-Davidson et al. 2001).

In 1990, a study in Germany analyzed factors that predicted people's exposures to VOCs and found that while smoking was the most significant determinant of benzene exposure, automobile-related activities, such as refueling and driving, were also significant (Hoffmann et al. 2000). Virtually all (99.9%) of the benzene released into the environment finally distributes itself into the air. The general population may be exposed to benzene through inhalation of contaminated air, particularly in areas of heavy motor vehicle traffic and around gas stations. Compared to inhalation, dermal exposure probably constitutes a minor portion of benzene exposure for the general population. Personal sources account for 18% of the total exposure of the general population to benzene. The main personal sources (other than smoking cigarettes) are driving or riding in automobiles and using products that emit benzene (paints, adhesives, marking pens, rubber products, and tapes) (Wallace 1989a).

Beginning in 2011, benzene in gasoline has been limited to 0.62% volume (EPA 2023a). Since benzene is a constituent of auto exhaust and fuel evaporation, people who spend more time in cars or in areas of heavy traffic have increased personal exposure to benzene. No recent U.S. monitoring data were located. Available exposure estimates and biomonitoring data were published prior to national benzene gasoline reductions and are based on historic levels of benzene in gasoline (1-–2%); these data likely overestimate exposures today. Assuming an average benzene concentration of 40 μg/m^3 (12.5 ppbv) for a moving automobile and an exposure duration of 1 hour/day, the calculated intake for driving or riding in an automobile in the late 1980s was 40 μg/day (Wallace 1989a). In an investigation of exposure to methyl *tert*-butyl ether (MTBE) in oxygenated gasoline in Stamford, Connecticut, venous blood samples were collected from 14 commuters and from 30 other persons who worked in the vicinity of traffic or automobiles. In addition to MTBE, the samples were analyzed for five chemicals, including benzene. Levels of benzene in the blood of 11 nonsmoking men and women commuters were 0.10–0.20 μg/L (median: 0.12 μg/L). Blood benzene levels of 0.29, 0.14, and 0.58 μg/L were measured in one female and two male smoking commuters, respectively (White et al. 1993).

Pumping gasoline can also be a significant source of exposure to benzene; these studies may be based on historic levels of benzene in gas and therefore overestimate exposures today. A study conducted between July 1998 and March 1999 that comprised of 39 customers of gasoline self-service stations from North Carolina, measured the benzene level in the air around the station as well as the levels of benzene in customers' breath prior to and immediately after fueling (Egeghy et al. 2000). Benzene levels in the air around the station were <0.02–11.16 ppmv (<0.06–35.65 mg/m^3), with a mean (±1 SD) of 0.91 (±1.8) ppmv (2.9±5.6 mg/m^3). The range of benzene levels in the breath of customers prior to fueling was <0.001–0.022 ppmv (<0.003–0.070 mg/m^3) with a mean (±1 SD) of 0.0027 (±0.0034) ppmv (0.0090±0.0110 mg/m^3) while the range of benzene levels in the breath of customers after re-fueling was <0.001–0.434 ppmv (<0.003–1.37 mg/m^3) with a mean (±1 SD) of 0.05 (±0.081) ppmv (0.16±0.259 mg/m^3) (Egeghy et al. 2000). Another study reported a benzene concentration of 1 ppm at the breathing-level of a person pumping gas (Bond et al. 1986b). Using this concentration and an estimated 70 minutes/year of time spent pumping gasoline, a benzene intake of 10 μg/day has been calculated (Wallace 1989a). In a group of 26 subjects who were not occupationally exposed to benzene, but were exposed to benzene during refueling in Fairbanks, Alaska, median blood benzene levels prior to and immediately following refueling were 0.19 ppbv (0.61 μg/m^3; range: 0.08–0.65 ppbv; 0.26–2.1 μg/m^3) and 0.54 ppbv (1.7 μg/m^3; range: 0.13–1.70 ppbv; 0.41–5.4 μg/m^3), respectively (Backer et al. 1997). While most human exposure to benzene is believed to be through inhalation, studies show that benzene can permeate skin with a permeability factor of about 0.14–0.18 cm/hour at 25°C. The permeability factor was not affected by moisturizer, baby oil, or insect repellent; however, it was affected by temperature (50°C) and sunscreen, with the permeability factors increasing to 0.26 and 0.24 cm/hour, respectively (Nakai et al. 1997).

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Children can be subject to increased benzene exposure by inhalation of second-hand smoke. In a study of nonsmoking rural families, urban families, and urban smoking families, infant exposure to benzene was estimated at doses of 15.3, 19.7, and 25.9 μg/day, respectively, with daily intakes of 1.68, 2.16, and 2.55 μg/kg body weight/day, respectively. For children of the same classification, benzene exposure was measured at doses of 29.3, 37.6, and 49.3 μg/day, respectively, with daily intakes of 0.71, 0.91, and 1.20 μg/kg bodyweight/day, respectively. For all infants and children, benzene exposure predominantly comes from the indoors (Duarte-Davidson et al. 2001).

Depending on the children's living environment, they may have higher exposure to benzene than adults. In a study of two lower-income areas of Minneapolis, children were found to have average personal benzene exposures of 0.66 and 0.53 ppb (2.1 and 1.7 μg/m^3) in the winter and spring, respectively. The highest concentration of benzene in their environment came from the home, with winter and spring concentrations of 0.69 and 0.66 ppbv (2.2 and 2.1 μg/m^3), respectively, while the outdoor and school benzene concentrations were 0.41 and 0.19 ppbv (1.3 and 0.60 μg/m^3), respectively (Adgate et al. 2004). In a follow-up study of Minneapolis lower-income children between 2000 and 2002, blood benzene was detected in 71.4% samples at a median of 0.08 ng/mL (range: 0.04–0.26 ng/mL) (Sexton et al. 2006). In Italy, concentrations of the benzene metabolite*, trans,trans*-muconic acid, were measured in the urine of children from both urban areas in Naples and rural areas in Pollica. The mean urinary concentrations of *trans,trans*-muconic acid detected for rural and urban children were 48.4 and 98.7 μg/L, respectively (Amodio-Cocchieri et al. 2001). These studies also found no strong link between passive smoking and *trans,trans*-muconic acid levels. The only factor that affected levels of *trans,trans*-muconic acid in urine samples was how close the family lived to the road. A study in Rouen, France, compared benzene exposure and concentrations in nonsmoking parents and their children. Despite the fact that the children were exposed to slightly less benzene (3.47 ppbv [11.1 μg/m^3]) than their parents (4.51 ppbv [14.4 μg/m^3]), there was no significant correlation between exposure means and urinary metabolite levels (Kouniali et al. 2003).

Lagorio et al. (2013) conducted a study in Italy assessing the exposure of benzene to children by repeated weekly measurements in breathing zone and ambient outdoor air samples along with determination of cotinine, *trans,trans*-muconic acid, and PhMA in urine. In 108 children, all between the ages of 2 and 12 years, the average benzene concentrations in personal and outdoor air samples were reported as 3 and 2.7 μg/m^3 (0.92 and 0.81 ppbv), respectively. The average urinary cotinine, *trans,trans*-muconic acid, and PhMA concentrations were 3.73, 116.65, and 1.28 μg/g creatinine, respectively.

Benzene in breast milk has been a major concern. Benzene was detected in U.S. breast milk samples from nonsmoking homes, at an average of 0.12 ng/mL (Kim et al. 2007b). Breast milk concentrations trended with indoor air concentrations. While this may provide a mechanism by which infants are exposed to benzene, these concentrations are lower than in other foods.

Individuals who live near hazardous waste sites, leaking underground fuel storage tanks, or oil natural gas drilling might be exposed to potentially high concentrations of benzene in their drinking water if they obtain tap water from wells located near these sources. Benzene was detected at a maximum of 7.65 ppb in drinking water sourced from groundwater impacted by natural gas drilling (ATSDR 2019b). In a series of experiments conducted in a single-family residence from June 11 to 13, 1991, exposure to benzene through contaminated residential water was monitored (Lindstrom et al. 1994). The residential water was contaminated with benzene and other hydrocarbons in 1986. Periodic testing conducted from 1986 to 1991 showed benzene concentrations of 33–673 μg/L (ppb). The experiment involved an individual taking a 20-minute shower with the bathroom door closed, followed by 5 minutes for drying and dressing; then the bathroom door was opened and the individual was allowed to leave the house. Integrated 60- and 240-minute whole-air samples were collected from the bathroom, an adjacent bedroom, the living room, and in ambient air. Glass, gas-tight syringe grab samples were simultaneously collected from the shower, bathroom, bedroom, and living room at 0, 10, 18, 20, 25, 25.5, and 30 minutes. Two members of the monitoring team were measured for 6 hours using personal Tenax gas chromatographic (GC) monitors. For the first 30 minutes of each experiment, one member was based in the bathroom and the other was based in the living room. Benzene concentrations in the shower head were 185–367 μg/L (ppb), while drain level samples ranged from below the detectable limit (0.6 μg/L or ppb) to 198 μg/L (ppb). Analysis of the syringe samples suggested a pulse of benzene moving from the shower stall to the rest of the house over approximately 60 minutes. Peak benzene levels were measured in the shower stall at 18–20 minutes (758–1,670 μg/m^3), in the bathroom at 10–25 minutes (366–498 μg/m^3), in the bedroom at 25.5–30 minutes (81–146 μg/m^3), and in the living room at 36–70 minutes (40–62 μg/m^3). The total benzene dose resulting from the shower was estimated to be approximately 281 μg, with 40% via inhalation and 60% via the dermal pathway (Lindstrom et al. 1994).

The major source of exposure to benzene is cigarette smoke. A smoker of 32 cigarettes per day would have a benzene intake of approximately 1.8 mg/day (at least 10 times the average nonsmoker's intake) (Wallace 1989a). Median benzene concentrations in 343 homes with smokers averaged 3.3 ppb (11 μg/m^3) compared to 2.2 ppb (7.0 μg/m^3) in 185 homes without smokers. This represents a 50% increase in benzene exposure for nonsmokers exposed to passive smoke compared to nonsmokers not exposed to passive smoke (Wallace 1989a). In a study in Germany, the mean benzene concentrations for frequent smokers and nonsmokers were 6.1 and 2.4 ppb (19 and 7.7 μg/m^3), respectively (Hoffmann et al. 2000).

In a study measuring the mean benzene exposure by monitoring urinary benzene excretion in 33 petrochemical plant workers, 30 small town residents 2 km from the plant, 26 small town residents 2–4 km from the plant, and 54 urban residents 25 km from the plant, nonsmokers had median urinary benzene concentrations of 236, 48, 63, and 120 ng/L, respectively, while smokers had median concentrations of 692, 470, 421, and 1,090 ng/L, respectively (Fustinoni et al. 2012).

A study compared the urinary benzene metabolite, *trans,trans*-muconic acid, between smokers and nonsmokers among 81 ceramic factory workers exposed to low levels of benzene and 83 general population controls (Ibrahim et al. 2014). Among the factory workers, 26 smokers had an average urinary *trans,trans*-muconic acid concentration of 0.252 mg/g creatinine, while 55 nonsmokers had an average of 0.183 mg/g creatinine. In the nonexposed control group, 25 smokers had an average urinary *trans,trans*-muconic acid concentration of 0.043 mg/g creatinine, while 58 nonsmokers had an average of 0.035 mg/g creatinine.

Individuals employed in industries that use or make benzene or products containing benzene may be exposed to the highest concentrations of benzene. The National Occupational Exposure Survey (NOES), conducted by the National Institute for Occupational Safety and Health (NIOSH) from 1981 to 1983, estimated that approximately 272,300 workers employed in various professions were potentially exposed to benzene in the United States. Approximately half of these workers were employed in general medical and surgical hospitals, and their occupations included nurses and aides, physicians, technicians, technologists, therapists, dieticians, pharmacists, and janitors (NIOSH 1990). The NOES database does not contain information on the frequency, concentration, or duration of exposure; the survey provides only estimates of workers potentially exposed to chemicals in the workplace. The current OSHA permissible exposure limit (PEL) for an 8-hour TWA exposure to benzene is 1 ppm (3 mg/m^3) and the short-term exposure limit (STEL) in any 15-minute period is 5 ppm (16 mg/m^3) (OSHA 2003). NIOSH recommended exposure limit is 0.1 ppm (0.3 mg/m^3) for a 10-hour workday during a 40-hour workweek and the short-term exposure is 1 ppm (3 mg/m^3) (NIOSH 1992). In 1987, OSHA estimated that approximately 238,000 workers were exposed to benzene in seven major industry sectors, including petrochemical plants, petroleum refineries, coke and coal chemicals, tire manufacturers, bulk terminals, bulk plants, and transportation via tank trucks (see Table 5-18) (OSHA 1987). Approximately 10,000 workers were estimated to be exposed to TWA concentrations in excess of the 1 ppm standard. This estimate did not include firms covered by the exclusions, firms under jurisdiction of other agencies, or firms involved in the use of products containing small quantities of benzene. The uptake of benzene by workers in a municipal waste incinerator in Germany was assessed by measuring benzene levels in blood

(Angerer et al. 1991). No significant difference ($p<0.05$) in blood benzene levels between workers and controls were detected (mean 0.22 μg/L for nonsmoking workers versus 0.25 μg/L for nonsmoking controls). OSHA requires the use of engineering controls and/or respiratory protection in situations where compliance with the TWA is not feasible (OSHA 1987).

Table 5-18. Percentage of Employees Exposed to Benzene by Exposure Level and Industry Division (1987)

Industry sector	Percentage of observations in each exposure category according to range of 8-hour TWA benzene concentrations (ppm)						Total number of employees
	0.0–0.1	0.11–0.5	0.51–1.0	1.1–5.0	5.1–10	10+	
Petrochemical plants[a]		74.6		23.0	2.4	0.0	4,300
Petroleum refineries[b,c]	64.6	26.1	4.6	3.8	0.5	0.4	47,547
Coke and coal chemicals[a]	0.0	39.3	27.6	27.5	4.4	1.3	947[d]
Tire manufacturers[b]	53.4	37.5	6.3	2.8	0.0	0.0	65,000
Bulk terminals[b]	57.8	32.8	5.3	3.7	0.3	0.1	27,095
Bulk plants[b]	57.8	32.8	5.3	3.7	0.3	0.1	45,323
Transportation via tank truck[b]	68.4	23.1	5.3	2.9	0.1	0.2	47,600
Total							237,812

[a] Percentages represent the portion of workers whose average exposures are in each category.
[b] Percentages represent the portion of sampling results in each category.
[c] Data do not reflect respirator use and sampling biases.
[d] Excludes workers employed at the coke ovens.

TWA = time-weighted average

Source: OSHA 1987

Certain jobs, such as gasoline station workers, firefighters, and drycleaners, are believed to put people at a higher risk of benzene exposure. In an analysis of literature, it was estimated that workers in the area of crude petroleum and natural gas are exposed to 0.04 ppmv (0.128 mg/m^3) benzene, while workers in petroleum refining, gas stations, and crude petroleum pipelines are exposed to 0.22, 0.12 and 0.25 ppmv benzene (0.70, 0.38, and 0.80 mg/m^3), respectively. This study also showed that firefighters are exposed to an average of 0.38 ppmv (1.2 mg/m^3) benzene (van Wijngaarden and Stewart 2003). Workers from four different drycleaning facilities in Korea had mean benzene air concentrations of 2.7–3.2 ppbv (8.6–10 μg/m^3). Their exposure to benzene was dependent upon the type of solvent used for cleaning (Jo and Kim 2001). Benzene concentrations of 25.46 and 1,331.29 ppbv (81.33 and 4,253.05 μg/m^3) were found near the kiln and at the rotary line, respectively, inside a hazardous waste incinerator in Turkey (Bakoglu et al. 2004).

In 12 nonsmoking male car repair workers working in Stamford, Connecticut, blood benzene levels were 0.11–0.98 μg/L (median: 0.19 μg/L); in 8 smoking male car repair workers, benzene levels were 0.17–0.67 μg/L (median: 0.42 μg/L). Three nonsmoking male gasoline attendants had blood benzene levels of 0.32–0.47 μg/L (median: 0.36 μg/L) (White et al. 1993).

A study comparing workers who were exposed to benzene regularly at work to people who were not exposed to benzene at work showed that while the general population in Italy had average blood benzene concentration of 165 ng/L, the people who were exposed to high benzene levels at work had an average benzene blood concentration of 186 ng/L. Immediately following their shift, the average benzene blood level samples from of benzene-exposed workers was 420 ng/L. The average blood benzene levels for smoking and nonsmoking occupationally exposed workers were 264 and 123 ng/L, respectively (Brugnone et al. 1998).

Bogen and Sheehan (2014) estimated that workers' dermal exposure to benzene in mineral spirits solvents (MSS), used in parts washing and degreasing operations, averaged 33% of their total (dermal and inhalation) benzene uptake. The estimated average benzene doses from parts washing by dermal exposure and total exposure were reported as 0.0093 and 0.054 mg/day, respectively, using 'low-aromatic' MSS formulations from 1995 to 1999. In a study assessing the dermal exposure and absorption of combustion contaminants in firefighters during six different controlled structure burns, the median increase in breath concentrations of benzene post- versus pre-burn were 48.1, 2.81, 39.2, -0.33, 7.39, and 18.8 μg/m^3 (15.1, 0.880, 12.2, -0.10, 2.31, and 5.88 ppbv) (Fent et al. 2014). The benzene metabolite, PhMA, could not be detected in urine samples collected from the firefighters (minimum detectable concentration 8.5 μg/g).

In a study comparing the urinary benzene metabolite *trans,trans*-muconic acid between 81 ceramic factory workers exposed to low levels of benzene and 83 nonexposed controls, the workers and the control group had mean *trans,trans*-muconic acid concentrations of 0.22 and 0.043 mg/g creatinine, respectively (Ibrahim et al. 2014).

One study assessed the exposure of 133 male petrochemical industry operators to benzene by both environmental (personal air) and biological monitoring (metabolites *trans,trans*-muconic acid and PhMA in end-shift urine). The mean reported values of benzene exposure were 0.014 ppm, 101 μg/g creatinine, and 2.8 μg/g creatinine for benzene, *trans,trans*-muconic acid, and PhMA, respectively (Carrieri et al.

2010). Another study assessed the occupational exposure of urban and rural female workers to benzene, toluene, and xylenes by monitoring urban air for traffic policewomen (street) versus police drivers (vehicle); monitoring urban air versus rural air; and biological monitoring of workers in urban areas versus rural areas (Ciarrocca et al. 2012). Mean personal air exposures to benzene over an 8-hour sampling period were similar for urban street (16.7 μg/m^3) and vehicle workers (18.7 μg/m^3), but were reported to be higher when compared to rural workers (less than the LOD of 1.6 μg/m^3). Mean blood and urine levels of benzene, and *trans,trans*-muconic acid and PhMA, respectively, were similar among the street (244.4 ng/L, 62.0 μg/g creatinine, 3.5 μg/g creatinine) and vehicle workers (241.1 ng/L, 61.8 μg/g creatinine, 3.4 μg/g creatinine), but blood levels of benzene were higher in urban workers compared to rural workers (113.1 ng/L, 40.8 μg/g creatinine, 2.8 μg/g creatinine) (Ciarrocca et al. 2012). In a study measuring the mean benzene exposure in 33 petrochemical plant workers, 30 small town residents 2 km from the plant, 26 small town residents 2–4 km from the plant, and 54 urban residents 25 km from the plant, measured median personal air benzene concentrations were 25, 9, 7, and 6 μg/m^3, respectively, while median urinary metabolite concentrations were 236, 48, 63, and 120 ng/L and 692, 470, 421, and 1,090 ng/L, for nonsmokers and smokers, respectively (Fustinoni et al. 2012).

A study determined benzene exposure in 33 petrochemical industry operators (PIOs), 28 service station attendants (SSAs), 21 gasoline pump maintenance workers (GPMWs), and 51 nonexposed controls by measuring personal air concentrations and benzene metabolites, *trans,trans*-muconic acid and PhMA, in end-of-shift urine samples (Fracasso et al. 2010). The levels of benzene (in μg/m^3) in personal air for PIOs, SSAs, GPMWs, and controls were 1.7–593.50 (median 27.8), 8.00–260.00 (median 40.00), 4.60–514.90 (median 24.20), and 1.97–16.3 (median 5.40), respectively. Urinary levels of metabolites (in μg/g creatinine), *trans,trans*-muconic acid and PhMA, in PIOs, SSAs, GPMWs, and controls were 49.00–422.00 (median 128.00) and 0.40–35.60 (median 8.60), 30.00–418.00 (median 117.00) and 1.55–15.00 (median 5.55), 13.40–242.50 (median 92.00) and 0.21–10.53 (median 1.77), and 3.00–460.50 (median 84.00) and 0.30–10.08 (median 1.90), respectively. The results show that in all groups of workers, the level of personal air exposure to benzene was higher than the control groups, while the level of urinary metabolites was higher in the SSA and PIO groups compared to the control. No increase in urinary metabolites was measured in GPMWs, but it was noted that for these workers, benzene exposure was not continuous and only occurred on specific days.