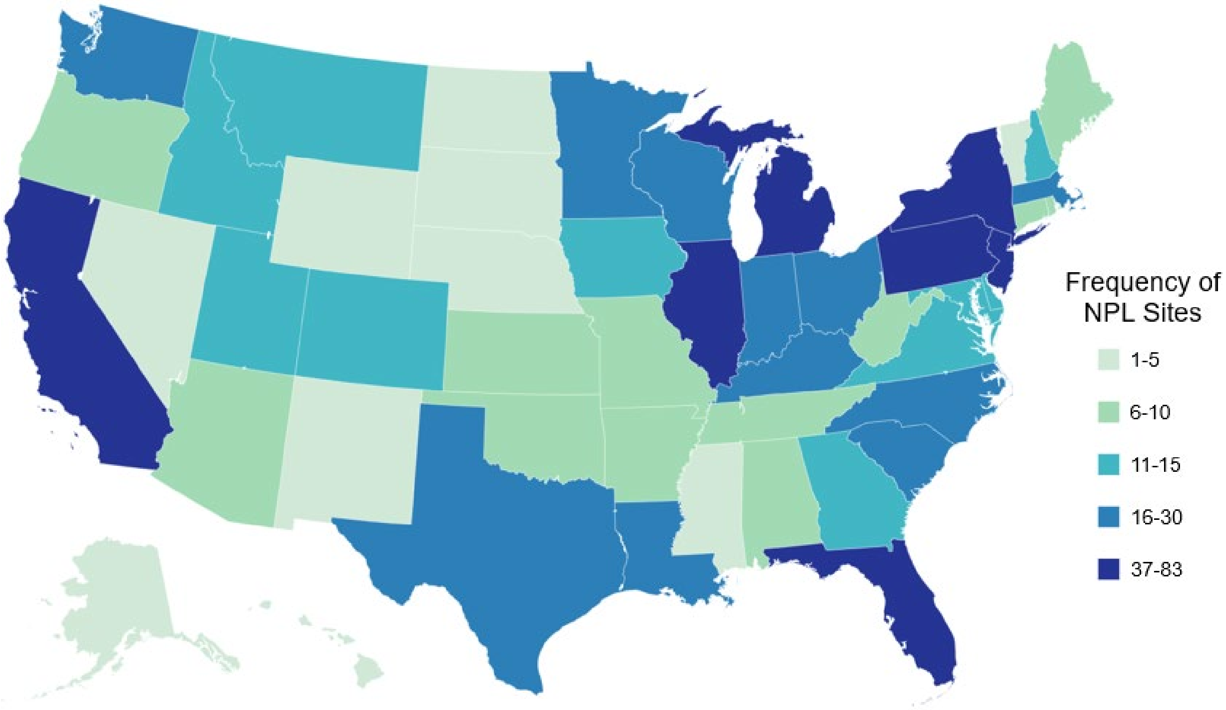
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Mercury or mercury compounds have been identified in at least 847 of the 1,867 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2022). However, the number of sites in which mercury has been evaluated is not known. The number of sites in each state is shown in Figure 5-1. Of these sites, 837 are located within the United States, 1 is located in Guam, 2 are located in the Virgin Islands, and 7 are located in Puerto Rico (not shown).

Figure 5-1. Number of NPL Sites with Mercury or Mercury Compound Contamination



Source: ATSDR 2022

- The general population is primarily exposed to mercury through the ingestion of foods, particularly fish.
- The general population may also be exposed to mercury by inhalation of ambient air. Exposure from ingestion of drinking water is a minor exposure pathway. Mercury released from mercury amalgam restorations can also contribute to mercury exposure.
- Occupational exposure for persons working with mercury or mercury compounds, such as mercury recycling and reprocessing facilities or dental offices where mercury is used in dental amalgams, may occur through inhalation or dermal contact.

- Most mercury in the atmosphere is in the gaseous elemental form, which can remain suspended in air for long periods of time and is subject to long-range atmospheric transport. When released or deposited to land or water, mercury can be transformed to methylated forms by anaerobic microorganisms. Methylmercury is highly bioaccumulative and biomagnifies in the food chain.

Mercury occurs naturally as a mineral and is distributed throughout the environment by both natural and anthropogenic processes. The natural global bio-geochemical cycling of mercury is characterized by degassing of the element from soils and surface waters, followed by atmospheric transport, deposition of mercury back to land, vegetation and surface water, and sorption of the compound to soil or sediment particulates. Mercury deposited on land and open water is, in part, revolatilized back into the atmosphere. This emission, deposition, and revolatilization creates difficulties in tracing the movement of mercury to its sources. Major anthropogenic sources of mercury releases to the environment include mining and smelting; industrial processes involving the use of mercury, including chloralkali production facilities; combustion of fossil fuels, primarily coal; production of cement; and medical and municipal waste incinerators and industrial/commercial boilers. Natural sources include volcanic activity, wildfires that release sequestered mercury from biomass, and weathering of mercury-containing rocks.

The element has three valence states and is found in the environment in the metallic form and in the form of various inorganic and organic complexes. The major features of the bio-geochemical cycle of mercury include degassing of mineral mercury from the lithosphere and hydrosphere, long-range transport in the atmosphere, wet and dry deposition to land and surface water, sorption to soil and sediment particulates, revolatilization from land and surface water, and bioaccumulation in both terrestrial and aquatic food chains.

Potential sources of general population exposure to mercury include inhalation of mercury vapors in ambient air, ingestion of drinking water and foodstuffs contaminated with mercury, and exposure to mercury through dental and medical treatments. Dietary intake is the most important source of nonoccupational exposure to mercury, with fish and other seafood products being the dominant source of mercury in the diet. Most of the mercury consumed in fish or other seafood is the highly absorbable methylmercury form. Consumption of rice can also make a substantial contribution to dietary mercury intake. Intake of elemental mercury from dental amalgams is another important contributing source to the total mercury body burden in humans in the general population. This is expected to decline as use of dental amalgams is being phased-out in many countries.

Mercury is present in a variety of human tissues. Mercury has also been detected in urine, human breast milk, nails, hair, and placenta in individuals in the general population. Inhalation of mercury vapor in workplace atmospheres is the main route of occupational exposure to the compound. Mercury has a long history of use in industrial processes and as a therapeutic agent (Clarkson and Magos 2006). Occupational exposure to mercury has occurred in a variety of industries that process or use the element (e.g., felting, chloralkali processing, fluorescent lamp production, gold mining and processing, lithium-6 purification, dentistry applications of mercury amalgam, mercury battery production, natural gas production, recycling, and thermometer production).

Members of the general public with potentially high exposures include individuals who live in proximity to former mercury mining or production sites, secondary production (recycling) facilities, municipal or medical incinerators, or coal-fired power plants. Other populations at risk of exposure include recreational and subsistence fishers who routinely consume meals of fish that may be contaminated; subsistence hunters who routinely consume the meat and organ tissues of marine mammals; individuals with a large number of dental amalgams; fetuses by maternal-fetal transfer and breastfed infants through maternal-breast milk transfer; medical exposure (e.g., ethylmercury used as a preservative in vaccines); occupational sources; individuals who use consumer products containing mercury (e.g., traditional or herbal remedies, or cosmetics, including skin lightening creams); and individuals where intentional (religious or cultural use) or unintentional mercury spills have occurred. Historically, mercury compounds were also used as pharmaceutical agents (e.g., antibiotics, antiseptics, diuretics) (Clarkson and Magos 2006).

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

Mercury is a naturally occurring element that is usually found as mercuric sulfide (cinnabar), an insoluble, stable compound. It occurs in soils at a concentration of approximately 80 ng/g (0.080 ppm) but the actual levels in different locations can vary considerably (Gonzalez-Raymat et al. 2017). Mercury is mined using both open pit (10% of production) and underground mining techniques (90%) (Drake 1981).

Mercury ores are processed inexpensively to produce metallic mercury. Due to the low boiling point of elemental mercury, mercury can be refined by heating the ore and condensing the vapor to form metallic mercury. This method is 95% efficient and yields mercury that is 99.9% pure. The methods used to refine mercury ores are uncomplicated. Smaller refineries use simple firing and condensing equipment, while larger operations use continuous rotary kilns or mechanically feeding and discharging multiple-hearth furnaces (DOI 1985).

According to the U.S. Geological Survey (USGS), mercury has not been produced as a principal mineral commodity in the United States since 1992; however, it is recovered as a byproduct from processing gold-silver ore at mines located in Nevada (USGS 2023a). Metals in the gold ores are extracted with an aqueous cyanide solution, with typical mercury recoveries of between 10 and 20% (DOI 1993; USGS 1997). In addition, mercury can be recovered from batteries, compact and traditional fluorescent lamps, dental amalgam, medical devices, old thermostats, and mercury-contaminated soils. It was estimated that in 2019, <40 metric tons of mercury were consumed domestically in the United States (USGS 2020). In 2021, the reported domestic consumption of mercury and mercury in compounds in products was 16 metric tons (USGS 2023a). The USGS reported that, in 2022, eight facilities operated by six companies in the United States accounted for the majority of secondary mercury produced and were authorized by the U.S. Department of Energy to temporarily store mercury (USGS 2023a). Typically, mercury-containing products such as automobile convenience switches, barometers, mercury containing lightbulbs, computers, dental amalgams, medical devices, and thermostats are retrieved by smaller companies and transported to the refining establishments for mercury reclamation. Due to the continued phase-out of compact and traditional fluorescent lighting for light-emitting-diode (LED) lighting, mercury recycling has increased.

Annual global mine production of mercury was estimated to be around 4,000 metric tons in 2019 (USGS 2020) and 2,200 metric tons in 2020 and 2021 (USGS 2023a). China is the overwhelming producer of mined mercury (~2,000 metric tons in 2020). Other nations with mine production of mercury include Argentina, Kyrgyzstan, Mexico, Norway, Peru, and Tajikistan (USGS 2023a). Gold mining may produce mercury. In 1995, there were eight U.S. mines that produced mercury when aqueous cyanide solution was used to recover metals (DOI 1993; USGS 1997). It is unclear whether this production occurs today as production volumes for this use are no longer disclosed.

Tables 5-1 and 5-2 summarize information on companies that reported the production, import, or use of elemental mercury and mercury compounds, respectively, for the Toxics Release Inventory (TRI) in 2021 (TRI22 2024). TRI data should be used with caution since only certain types of industrial facilities are required to report. This is not an exhaustive list.

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

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AK	4	0	99	1, 5, 12, 14
AL	8	0	999	1, 5, 8, 12, 13, 14
AR	6	0	99	1, 5, 12, 13, 14
AZ	11	0	9,999	1, 4, 5, 6, 8, 9, 12, 13, 14
CA	31	0	9,999	1, 2, 5, 8, 10, 11, 12, 13, 14
CO	6	0	99	1, 5, 12, 13, 14
CT	5	0	999	6, 7, 8, 9, 12, 14
DC	1	0	99	14
DE	1	100	999	8
FL	3	0	9,999	1, 4, 5, 8, 12, 13, 14
GA	8	0	999	1, 2, 3, 4, 5, 6, 8, 12, 13, 14
GU	1	0	99	1, 5, 12
IA	6	0	99	1, 5, 8, 12, 13, 14
ID	2	1,000	9,999	1, 5, 12, 13, 14
IL	23	0	9,999	1, 5, 6, 9, 11, 12, 13, 14
IN	9	0	499,999,999	1, 2, 5, 8, 12, 13, 14
KS	3	0	999	1, 2, 5, 12, 13, 14
KY	5	0	99	7, 12, 14
LA	11	0	9,999,999	1, 2, 5, 12, 13, 14
MA	7	0	9,999	8, 9, 11, 12, 14
MD	4	0	999	11, 12, 14
ME	1	0	99	12
MI	6	0	9,999	1, 5, 8, 12, 13, 14
MN	13	0	9,999	1, 2, 5, 7, 8, 12, 13, 14
MO	5	0	99	1, 2, 5, 7, 9, 12, 13, 14
MS	3	0	99	1, 5, 9, 12, 14
MT	2	0	99	1, 10, 13
NC	9	0	999	1, 12, 13, 14
NE	7	0	9,999	1, 5, 12, 13, 14
NJ	4	0	9,999	8, 12, 14
NM	1	10,000	99,999	1, 5, 12
NV	9	0	9,999,999	1, 3, 5, 8, 9, 12, 14
NY	8	0	49,999,999	1, 5, 7, 11, 12, 14
OH	20	0	9,999	1, 2, 5, 7, 8, 12, 13, 14
OK	4	0	99	1, 2, 5, 8, 9, 12, 13, 14
OR	2	0	999	12, 14
PA	12	0	99,999	1, 2, 3, 4, 5, 8, 9, 12, 13, 14
RI	1	0	99	12, 14

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

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
SC	4	0	99	1, 5, 12, 14
SD	4	0	99	1, 5, 7, 12, 14
TN	5	0	99	1, 5, 12, 14
TX	38	0	9,999	1, 2, 4, 5, 7, 8, 9, 11, 12, 13, 14
UT	9	0	999	1, 5, 9, 12, 13, 14
VA	6	0	999	1, 5, 11, 12, 14
VT	1	0	99	8
WA	4	0	999	1, 5, 6, 8, 9, 11, 12
WI	7	0	999	1, 5, 14
WV	7	0	999	1, 5, 12, 13, 14
WY	5	0	9,999	1, 2, 5, 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 2024 (Data are from 2022)

Table 5-2. Facilities that Produce, Process, or Use Mercury Compounds

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	0	99,999	1, 2, 5, 7, 8, 9, 12, 13, 14
AL	29	0	999,999	1, 3, 4, 5, 6, 8, 12, 13, 14
AR	16	0	99,999	1, 2, 3, 4, 5, 6, 9, 11, 12, 13, 14
AZ	16	0	99,999	1, 3, 4, 5, 8, 9, 12, 13, 14
CA	63	0	99,999	1, 2, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
CO	18	0	9,999,999	1, 4, 5, 6, 7, 8, 12, 13, 14
CT	1	0	99	8, 12, 14
DC	1	0	99	14
DE	3	0	99	1, 2, 3, 5, 7, 12, 13, 14
FL	31	0	99,999	1, 2, 3, 4, 5, 8, 9, 11, 12, 13, 14
GA	18	0	999	1, 3, 4, 5, 7, 8, 12, 13, 14
GU	1	0	99	7, 9, 0
HI	6	0	99	1, 2, 5, 9, 13, 14
IA	20	0	9,999	1, 3, 4, 5, 7, 8, 9, 12, 13, 14

Table 5-2. Facilities that Produce, Process, or Use Mercury Compounds

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
ID	11	0	99,999	1, 5, 7, 8, 11, 12, 13, 14
IL	34	0	9,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
IN	34	0	9,999	1, 2, 3, 4, 5, 7, 9, 11, 12, 13, 14
KS	16	0	9,999	1, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
KY	25	0	9,999	1, 2, 3, 4, 5, 8, 9, 12, 13, 14
LA	44	0	99,999	1, 2, 3, 4, 5, 9, 11, 12, 13, 14
MA	1	100	999	2, 3, 4, 9, 0
MD	9	0	999	1, 3, 4, 5, 9, 12, 13, 14
ME	3	0	999	1, 5, 12, 14
MI	24	0	99,999	1, 4, 5, 7, 8, 9, 12, 13, 14
MN	18	0	9,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
MO	23	0	9,999	1, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
MP	1	0	99	7, 9, 0
MS	12	0	999	1, 2, 3, 4, 5, 8, 12, 13, 14
MT	13	0	999	1, 2, 5, 6, 12, 13, 14
NC	38	0	999,999	1, 2, 3, 5, 7, 8, 9, 12, 13, 14
ND	16	0	9,999	1, 2, 4, 5, 7, 9, 12, 13, 14
NE	14	0	99,999	1, 3, 4, 5, 7, 9, 12, 13, 14
NH	2	0	99	1, 2, 5, 9, 12, 13, 14
NJ	13	0	9,999	1, 2, 3, 4, 5, 7, 9, 12, 13, 14
NM	8	0	999	1, 3, 4, 5, 9, 12, 13, 14
NV	37	0	9,999,999	1, 2, 3, 5, 9, 12, 13, 14
NY	10	0	9,999	1, 5, 12, 13, 14
OH	35	0	9,999	1, 2, 3, 4, 5, 7, 9, 11, 12, 13, 14
OK	22	0	99,999	1, 2, 3, 4, 5, 8, 9, 12, 13, 14
OR	10	0	99	1, 2, 5, 8, 12, 13, 14
PA	50	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
PR	2	0	99	1, 2, 5, 12, 14
RI	2	0	999	7, 8, 14
SC	27	0	99,999	1, 3, 4, 5, 6, 7, 8, 12, 13, 14
SD	3	0	99,999	1, 5, 9, 12, 13, 14
TN	23	0	999	1, 2, 5, 8, 9, 12, 13, 14
TX	94	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
UT	27	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
VA	15	0	999	1, 3, 4, 5, 6, 7, 9, 12, 13, 14
WA	22	0	999	1, 2, 3, 5, 7, 8, 9, 12, 13, 14
WI	22	0	99,999	1, 3, 4, 5, 7, 8, 9, 12, 13, 14

Table 5-2. Facilities that Produce, Process, or Use Mercury Compounds

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
WV	16	0	999,999	1, 3, 4, 5, 6, 8, 9, 12, 13, 14
WY	13	0	999	1, 2, 3, 4, 5, 7, 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 2024 (Data are from 2022)

5.2.2 Import/Export

Until 1989, the United States was a net importer of mercury. After that, market values of mercury fluctuated and consumption diminished, leading to a decreased need for imported mercury (DOI 1985; Drake 1981). U.S. imports of mercury fell sharply between 1987 and 1990 (DOI 1990, 1993). The import volumes were: 636 metric tons in 1987, 329 metric tons in 1988, 131 metric tons in 1989, and 15 metric tons in 1990. However, imports of mercury began increasing after 1990: 56 metric tons in 1991, 92 metric tons in 1992, 40 metric tons in 1993, 129 metric tons in 1994, and 277 metric tons in 1995 (USGS 1997). Most recent data show low import volume as compared to the 1990s. In 2015, 2016, 2017, 2018, and 2019, mercury imports were reported as 26, 24, 20, 6, and 10 metric tons, respectively (USGS 2020). According to USGS (2020) for the period 2016–2018, imports were from Canada (39%), France (32%), Switzerland (13%), China (8%), and other countries (8%) (USGS 2020). From 2018 to 2021, most imports were from Canada (69%) and China (31%) (USGS 2023a).

Exports of elemental mercury were effectively eliminated on January 1, 2013, as a result of the Mercury Export Ban passed by Congress in 2008 (DeVito and Brooks 2013; EPA 2023). In addition, beginning on January 1, 2020, exports of five mercury compounds have also been banned (Mercury (I) chloride or calomel; mercury (II) oxide; mercury (II) sulfate; mercury (II) nitrate; and cinnabar or mercury sulphide) (EPA 2020b, 2023). In 2008, 2009, 2010, 2011, and 2012, U.S. exports of mercury were 732, 753, 459, 132, and 110 metric tons, respectively (USGS 2013).

5.2.3 Use

Mercury has many applications in industry due to its unique properties, such as its fluidity, its uniform volume expansion over the entire liquid temperature range, its high surface tension, and its ability to alloy with other metals. However, domestic consumption of mercury has shown a downward trend since the early 1970s. In 1995, consumption was 463 metric tons, down 10% from 1994. In 2019, consumption was estimated as <40 metric tons (USGS 2020). The EPA reported a revised domestic production of 45 metric tons in 2018, and about 82 metric tons of mercury was stored by manufacturers or producers. The reported domestic consumption of mercury in products was 16 metric tons (USGS 2023a).

The leading domestic end uses of mercury and mercury compounds were dental amalgam, 43%; relays, sensors, switches, and valves, 41%; bulbs, lamps, and lighting, 8%; formulated products (buffers, catalysts, fixatives, and vaccination uses), 7%; and batteries and other end uses, 1%. A large quantity of mercury (about 245 metric tons) is used domestically in manufacturing processes such as catalysts or as a cathode in the chlorine-caustic soda (chloralkali) process. Almost all the mercury is reused in the process (USGS 2023a).

In 2020, the use of mercury in the production of chloralkali decreased when one of the two operating facilities in the United States converted to a non-mercury process. Comparing data from 2018 to 2021, there has been an approximate 26% decrease in the amount of mercury sold in the United States and a >20% decrease in the amount of mercury used in products manufactured in the United States (EPA 2023). In 2021, 29,255 pounds of mercury were used for the domestic manufacture of mercury-added products (EPA 2023). Of this total, 27,276 pounds of elemental mercury were used to manufacture elemental mercury-added products and 1,979 pounds of mercury compounds were used to manufacture mercury compound-added products (EPA 2023).

Many past uses of mercury, such as in automobile convenience switches, alkaline batteries, paints and pigments, fungicides and pesticides, thermometers, and other scientific and medical devices, have been discontinued or significantly reduced, although these historical uses still represent a significant emission source (CDC 2015). For example, it has been estimated that 150–200 tons of mercury are still contained in old automobiles in the United States and up to 10 tons of mercury per year may be released from shredded vehicles (DeVito and Brooks 2013). Another historical use of note is the former use of mercury in the making of felt hats starting during the industrial revolution through the early 20th century (Beauchamp et al. 2021; Byrns and Pennings 2017). Hatters using this process were often exposed to high levels of mercury, which resulted in a characteristic neurological syndrome referred to "Mad Hatter's Disease" (Section 2.16).

Historically and even presently, gold mining uses mercury and may even produce it (Section 5.2.1). Amalgamation is the process where mercury is added to substrates containing gold to form a complex. The mercury-gold complex/amalgam is then heated to release the mercury, resulting in gold. There is industrial scale gold mining and artisanal and small-scale gold mining (ASGM). ASGM uses mercury to extract gold. Large-scale operations may use mercury and other processes in gold extraction, although ASGM can occur anywhere, the activity seems to be more predominant in developing countries. Populations in the Amazon River basin, some African countries, Slovenia, and India (Subhavana et al. 2019) are undertaking artisanal mining, as evidenced by some recent neurological studies of mercury exposed workers (Section 2.16.1 and Table 2-45).

Religious and Cultural Rituals, Ceremonies, and Practices. Most medicinal and pharmaceutical uses of mercury compounds have been discontinued. However, individuals in some cultural or religious groups may still use mercury in various rituals, practices, and ceremonies, resulting in exposure to elevated mercury concentrations in homes and confined spaces. Media reports (Sawyer 2015; Vasquez 2012; Washam 2011), case reports, and scientific papers indicate that mercury is still being used in this way and can cause health effects (Pandalai and Morgan 2011; Rhee et al. 2020; Tarabar and Su 2003; Weinstein and Bernstein 2003).

Metallic mercury has been used in Latin American and Caribbean communities as part of certain religious practices (e.g., Voodoo, Santeria, and Espiritismo), predominantly in domestic settings (Riley et al. 2001; Wendroff 1990; WHO 2010). Vietnamese and Indian people may also use mercury as alternative medicine. There are few instances in the literature where external or internal exposure measures have been quantified along with the health effect(s) observed. One case report details a Vietnamese person heating 'pellets' to vaporize the contents. After onset of malaise, dry skin, and poor appetite, the person saw a physician and blood tests indicated 409 μg mercury/L at 1-week postexposure. At 4 weeks postexposure, BHg was 61 μg/L, with 497 μg mercury/g creatinine in the urine (Pandalai and Morgan 2011). An unintentional poisoning occurred when 20-month-old Indian twins were given mercury 'teething powder' over the course of months. This resulted in the twins becoming weak; exhibiting a rash; and having swollen, red, painful hands and feet (these symptoms are consistent with acrodynia; see Section 2.15 for additional information). Testing indicated diminished reflexes with BHg levels of

176 and 209 µmol/L (Weinstein and Bernstein 2003). For more information about ritualistic mercury use, recommendations, and how the EPA has addressed this, please refer to EPA (2002b, 2006).

Electrical Applications. Mercury was a critical element in alkaline batteries; however, mercury use in batteries has been discontinued in the United States, with a few exceptions. The Mercury-Containing and Rechargeable Battery Management Act of 1996 began the phase out of mercury in batteries in the United States. Common AAA, AA, C, and D alkaline batteries no longer contain mercury. The total amount of mercury sold in batteries has declined from 5,585 pounds in 2001 to 1,203 pounds in 2013 (IMERC 2015). Currently, the only types of batteries in the United States that contain mercury are specialty button cell batteries and mercuric oxide batteries used for military and medical applications that that require a high-energy density and a flat voltage curve (IMERC 2015). The EPA estimated that, in 2018, approximately 8,915 pounds of mercury were used to produce switches, relays, and sensors in the United States and an additional 1,637 pounds of mercury were used in lighting lamps and bulbs (EPA 2020b). For 2021, it was estimated that 162 pounds were used for linear fluorescent lighting, lamps, and bulbs, with about 16 pounds exported for these purposes (EPA 2023). Another 16 pounds were used for high-pressure sodium and metal halide bulbs (EPA 2023). In 2021, 19,116 pounds of mercury were used in switches, relays, sensors, and bulbs (EPA 2023).

Medical Applications. Although many medical and laboratory uses of mercury are being phased out, mercury has been used domestically in laboratories and in a number of medical devices such as gastrointestinal dilators, manometers, sphygmomanometers, and thermometers (DeVito and Brooks 2013). It is also widely used in dental amalgam fillings, which contain approximately 50% metallic mercury, 35% silver, 9% tin, 6% copper, and trace amounts of zinc. The EPA reported that, in 2018, approximately 9,287 pounds of elemental mercury were used to produce dental amalgam in the United States (EPA 2020b). In 2021, that total was 7,995 pounds (EPA 2023). Thimerosal is a mercury-containing compound that prevents the growth of bacteria and fungus and is still used as a preservative for flu vaccines in multi-dose vials to keep the vaccine free from contamination (CDC 2011). It is no longer used in childhood vaccines. The last children’s vaccines that used thimerosal as a preservative expired in 2003 (CDC 2020). In 2018, it was estimated that approximately 290 pounds of mercury were used in vaccine usage in the United States (EPA 2020b). In 2021, 1,955 pounds were used in medical formulated products such as animal vaccines, preservatives, reagents, testing kits, etc. (EPA 2023).

Chemical/Mining Applications. Mercury is a catalyst in reactions to form polymers, such as vinyl chloride and urethane foams. The preparation of chlorine and caustic soda (NaOH) from brines also uses mercury as a catalyst. In this process, mercury is used as a moving cathode to separate sodium and chlorine (Rieber and Harris 1994). This mercury can be recycled with 95% efficiency (Drake 1981). Consumption occurs as mercury is lost in wastewater treatment, recaptured, reprocessed, and sent to landfills (Rieber and Harris 1994).

5.2.4 Disposal

Mercury is an element; thus, its chemical structure cannot be further broken down. In its elemental form, mercury is highly toxic when its vapors are inhaled. Therefore, incineration of mercury is not recommended as a disposal method. Mercury-containing waste products include waste effluents from chloralkali plants and discarded mercury-containing mechanical and electrical devices (DOI 1985). Under current federal guidelines, mercury and its compounds are considered hazardous substances, and various regulations are in effect to control the emission of mercury into the environment (especially organic compounds) (DOI 1985). Emissions from mercury ore processing facilities and mercury cell chloralkali plants are limited to 2.3 kg/day/facility. Emissions of mercury from the incineration or drying of wastewater sludges is limited to 3.2 kg/day/facility (EPA 1975a, 1975b). In addition, dumping wastes containing more than trace amounts of mercury is prohibited. There is currently no commercial capacity for disposal of high-concentration mercury containing hazardous waste. Management options in the United States for high-concentration mercury wastes of all types are limited. Despite the Department of Energy's (DOE) efforts to open and operate a long-term storage facility for elemental mercury, no such facility is currently available (EPA 2023); therefore, there are eight facilities operated by six companies in the United States that are authorized by the DOE to temporarily store mercury (USGS 2023a).

Recycling of mercury-containing compounds is an important method of disposal. Recycling (retorting) is a treatment for five categories of mercury wastes including: (D009) characteristic mercury; (K106) chlor-alkali waste; (P065) mercury fulminate; (P092) phenylmercuric acetate; and (U151) elemental mercury. From 1987 to 1991, annual production of mercury from old scrap averaged nearly 180 metric tons, equivalent to 16% of the average reported consumption during that period (DOI 1993). Virtually all mercury can be reclaimed from mercury cell chloralkali plants, electrical apparatuses, and control instruments when plants are dismantled or scrapped (DOI 1985). Increased recycling would decrease the mercury load from waste sites and treatment plants. As environmental concerns increase with respect to the disposal of mercury, the recovery by recycling and industrial processes will become a more significant source of domestic supply (DOI 1985).

5.3 RELEASES TO THE ENVIRONMENT

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

Mercury is released to the environment by both natural processes (e.g., volcanic activity, wildfires that release sequestered mercury from biomass, and weathering of mercury-containing rocks) and anthropogenic sources. Pyle and Mather (2003) analyzed data from active volcanoes and estimated that annual mercury emissions from volcanic activity was approximately 700 Mg/year (700 metric tons/year), which accounts for roughly 20–40% of all natural emissions. Studies of 11 western U.S. states between 2000 and 2013 estimated that the average annual emission of mercury due to wildfires in these states was 3,100±1,900 kg/year (3.1 metric tons/year) (Webster et al. 2016).

Anthropogenic releases have historically been primarily to the atmosphere; however, in the United States, these levels have been decreasing as regulations and engineering controls on point source and fugitive emissions limit the amount of mercury released to air. On-site land disposal now accounts for the majority of all of the releases of mercury from facilities that are required to report to the TRI (EPA 2020a). Streets et al. (2017) performed a comprehensive temporal review of worldwide anthropogenic emission sources of mercury and estimated that a cumulative total of 1,540 Gigagrams (1,540,000 metric tons) of mercury have been released to the environment from human activity, the bulk of which (24%) occurred from silver mining and smelting. The cumulative anthropogenic emission budget of mercury to the environment is shown in Table 5-3.

Table 5-3. Cumulative Worldwide Man-made Releases of Mercury to Air, Land, and Water Until 2010

Emission source	Amount released to air (Gg)	Amount released to land/water (Gg)	Total amount (Gg)
Copper smelting	4.91	12.6	17.5
Zinc smelting	10.5	25.3	35.8
Lead smelting	6.04	8.57	14.6
Iron making	1.2	1.45	2.65
Steel making	0.41	2.41	2.82
Mercury production	91.7	321	413
Gold, large-scale production	21	114	135
Gold, artisanal	34.4	51.6	86
Silver production	146	219	365
Cement production	3.29	3.47	6.76
Caustic soda production	8.80	63.6	72.4
Coal combustion	26.4	11.4	37.8
Oil combustion	0.77	0	0.77
Municipal waste incineration	34.6	0	34.6
Other waste burning	27.8	0	27.8
Electrical and measuring equipment	5.52	97.7	179
Chemicals manufacturing	47.5	131	179
Dental	1.06	5.69	6.75
Total Gg (metric tons)	472 (472,000)	1,070 (1,070,000)	1,540 (1,540,000)

Source: Adapted with permission from Streets et al. 2017. Copyright (2017) American Chemical Society.

5.3.1 Air

Estimated releases of 4,289 pounds (~1.95 metric tons) of elemental mercury to the atmosphere from 351 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for about 6.7% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). Estimated releases of 30,136 pounds (~13.67 metric tons) of mercury compounds to the atmosphere from 1,007 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for about 0.72% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). These releases are summarized in Tables 5-4 and 5-5.

Table 5-4. Releases to the Environment from Facilities that Produce, Process, or Use Elemental Mercury[a]

	Reported amounts released in pounds (metric tons) 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	8	710	0	0	266	0	964	12	976
AK	4	5	0	0	39	0	5	39	44
AZ	11	37	0	0	109	41	144	43	187
AR	6	59	0	0	209	0	268	0	268
CA	31	318	7	0	11,538	2	11,854	11	11,865
CO	6	21	0	0	0	0	21	0	21
CT	5	0	0	0	118	47	0	165	165
DE	1	0	0	0	0	0	0	0	0
DC	1	0	0	0	0	0	0	0	0
FL	3	56	0	0	5	0	56	5	61
GA	8	5	1	0	23	0	7	23	30
ID	2	1	0	0	2,834	2,538	2,835	2,538	5,373
IL	23	34	1	0	2,889	10	2,727	206	2,933
IN	9	89	19	0	109	0	195	22	218
IA	6	280	0	0	21	0	281	19	300
KS	3	50	0	1	18	2	51	20	71
KY	5	8	5	0	632	10	623	32	655
LA	11	673	160	0	2,703	361	3,456	442	3,898
ME	1	0	0	0	0	0	0	0	0
MD	4	0	0	0	0	25	0	25	25
MA	7	0	0	0	0	21	0	21	21
MI	6	86	0	143	9	0	238	0	238
MN	13	7	1	0	33	0	8	33	41
MS	3	4	0	0	1	0	4	1	4
MO	5	34	0	0	7	0	40	1	40
MT	2	1	0	0	0	0	1	0	1
NE	6	30	1	0	53	169	31	222	252
NV	9	271	0	0	3,453	20	3,724	20	3,744
NJ	4	1	1	0	90	0	2	90	92
NM	1	5	0	0	121	49	5	170	175
NY	8	38	5	0	802	13	838	20	859
NC	9	17	0	0	0	1	17	1	18
OH	20	738	36	0	614	91	1,334	144	1,478
OK	4	9	0	0	1	0	10	0	10
OR	2	0	0	0	415	325	415	325	741
PA	12	34	0	0	765	18,287	392	18,695	19,087
RI	1	0	0	0	0	0	0	1	1

Table 5-4. Releases to the Environment from Facilities that Produce, Process, or Use Elemental Mercury[a]

		Reported amounts released in pounds (metric tons) 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
SC	4	33	0	0	7	0	40	0	40
SD	4	41	0	0	2	0	42	0	42
TN	5	21	0	0	22	21	21	43	64
TX	38	145	5	1	4,288	921	4,419	941	5,360
UT	9	34	0	0	90	6	120	9	130
VT	1	0	0	0	0	1	0	1	1
VA	6	2	1	0	2	13	2	16	18
WA	4	3	0	0	14	5	16	6	22
WV	7	16	0	0	3,079	0	3,094	1	3,095
WI	7	12	0	0	242	10	12	252	264
WY	5	355	0	0	461	0	815	1	816
GU	1	6	0	0	0	0	6	0	6
Total	351	4,289	245	145	36,081	22,988	39,133	24,615	63,748

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

RF = reporting facilities; UI = underground injection

Source: TRI22 2024 (Data are from 2022)

Table 5-5. Releases to the Environment from Facilities that Produce, Process, or Use Mercury Compounds[a]

		Reported amounts released in pounds (metric tons) 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	29	1,636	17	0	126,456	329	127,698	740	128,438

Table 5-5. Releases to the Environment from Facilities that Produce, Process, or Use Mercury Compounds[a]

Reported amounts released in pounds (metric tons) 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
AK	18	62	0	0	92,634	61	92,686	72	92,758
AZ	16	561	2	0	40,562	5,693	36,499	10,318	46,818
AR	16	1,577	10	22	310,385	3	309,033	2,965	311,998
CA	59	663	52	0	26,625	1,276	25,931	2,685	28,616
CO	18	376	0	0	2,384	20	2,624	155	2,779
CT	1	0	0	0	1	0	1	0	1
DE	3	11	1	0	21	1	33	1	34
DC	1	5	0	0	0	0	5	0	5
FL	31	827	688	0	8,355	134	9,817	186	10,003
GA	18	209	29	0	342	8	536	52	588
HI	6	24	0	1	12	0	26	12	38
ID	11	740	6	0	7,165	0	7,910	1	7,910
IL	31	898	12	0	1,247	478	2,002	633	2,635
IN	33	3,792	11	0	2,211	139	5,804	348	6,153
IA	20	189	10	0	693	86	791	187	978
KS	16	226	1	0	608	201	813	223	1,036
KY	25	878	4	2	3,100	7	3,978	12	3,990
LA	42	513	18	17	1,980	352	1,744	1,136	2,879
ME	3	32	49	0	15	1	87	9	96
MD	9	140	0	0	424	0	140	424	564
MA	1	0	0	0	0	11	0	11	11
MI	23	275	2	0	3,045	117	3,194	244	3,438
MN	18	172	0	0	794	1	868	100	967
MS	12	388	15	29	464	22	605	313	918
MO	23	1,040	5	1	969	413	1,937	491	2,428
MT	12	163	0	0	1,301	10	1,441	33	1,474
NE	13	299	1	0	49,552	28	47,896	1,984	49,880
NV	33	1,876	0	5	3,075,211	284,349	3,076,115	285,327	3,361,441
NH	2	2	0	0	95	0	3	94	97
NJ	13	25	1,243	0	109	318	25	1,671	1,695
NM	8	84	0	1	1,117	350	1,201	351	1,552
NY	10	84	0	0	2,464	2	2,528	23	2,551
NC	38	914	12	0	4,342	93	2,105	3,256	5,361
ND	15	925	1	0	564	41	1,335	196	1,531
OH	35	1,006	47	166	3,223	65	3,297	1,210	4,507
OK	21	323	2	0	887	529	1,169	572	1,741
OR	10	54	4	0	2,309	0	59	2,308	2,366

Table 5-5. Releases to the Environment from Facilities that Produce, Process, or Use Mercury Compounds[a]

Reported amounts released in pounds (metric tons) 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
PA	50	1,295	20	0	2,600	885	2,459	2,342	4,801
RI	2	6	0	0	0	0	6	0	6
SC	27	1,298	11	0	938	110	2,186	171	2,357
SD	3	11	0	0	4,640	0	4,650	0	4,650
TN	23	704	80	0	1,827	226	2,255	582	2,837
TX	93	3,617	19	47	13,017	903	15,364	2,239	17,604
UT	25	628	200	0	57,450	96	52,516	5,858	58,375
VA	15	333	6	23	348	0	660	49	709
WA	22	167	2	0	158	14	279	63	342
WV	16	441	8	0	7,470	19	5,065	2,873	7,938
WI	21	168	145	0	978	171	900	561	1,460
WY	13	355	0	0	637	9	807	194	1,001
GU	1	0	0	0	0	0	0	0	0
MP	1	0	0	0	0	0	0	0	0
PR	2	126	0	0	192	0	126	192	318
Total	1,007	30,136	2,735	315	3,861,917	297,572	3,859,208	333,467	4,192,675

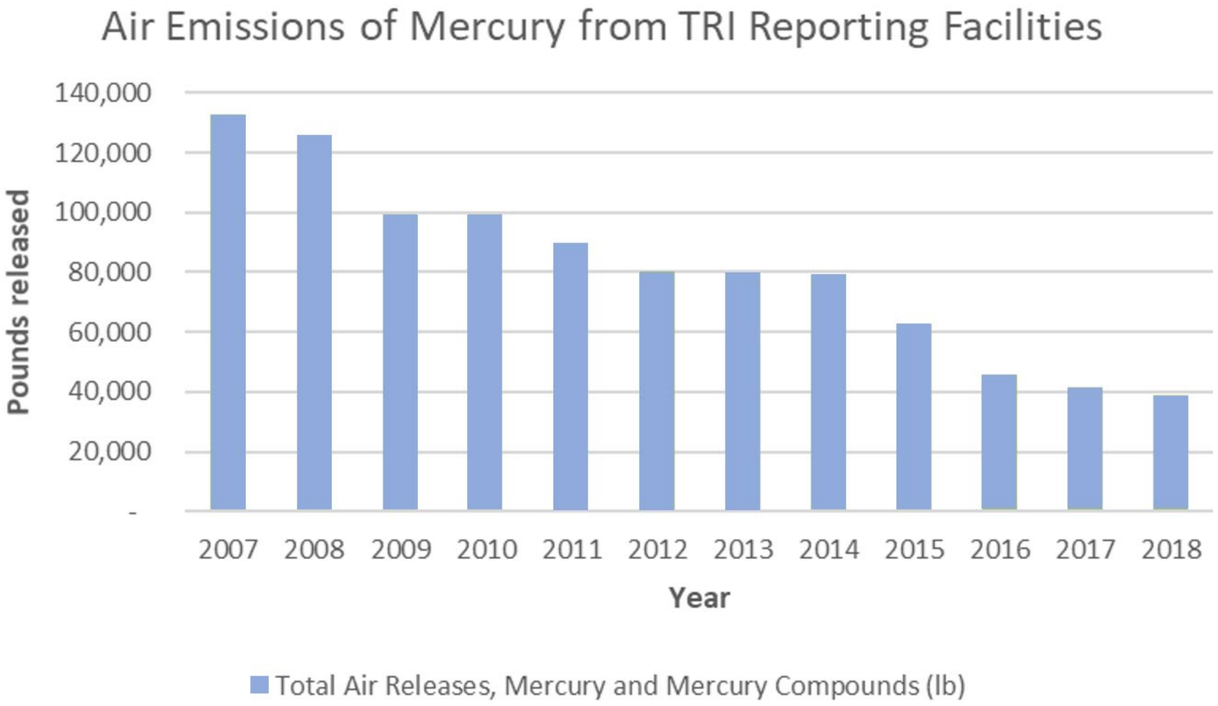
[a] The TRI data should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list. Data are rounded to nearest whole number.
[b] Data in TRI are maximum amounts released by each facility.
[c] Post office state abbreviations are used.
[d] Number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

RF = reporting facilities; UI = underground injection

Source: TRI22 2024 (Data are from 2022)

In the United States, atmospheric releases of mercury have been declining for the last 2 decades as fugitive and stack source emissions have been reduced. Figure 5-2 shows the atmospheric emissions of facilities required to report to the TRI from 2007 to 2018 (EPA 2020a).

Figure 5-2. Temporal Atmospheric Emissions of Mercury from Facilities Required to Report to the Toxics Release Inventory



Source EPA 2020a

The bulk of these emissions (>90%) arise from stack emissions rather than fugitive emissions. Releases of mercury and mercury compounds to air decreased by 71% from 2007 to 2018, with electric utility providers having a decline in mercury air emissions of approximately 90% during this period (EPA 2020b).

The United Nations Global Mercury Assessment for 2018 estimated that the global inventory of mercury emissions to the atmosphere from anthropogenic sources in 2015 was approximately 2,220 metric tons (UNEP 2018), which is approximately 20% greater than in 2010 (Dastoor et al. 2022). Artisanal and small-scale gold mining is the greatest emission source, representing approximately 38% of the total followed by stationary combustion of coal from power plants (13.1%), cement production (10.5%), and non-ferrous metal production, principally aluminum, copper, lead, and zinc (10.3%). Anthropogenic emissions by sector source type are illustrated in Table 5-6. Globally, the greatest amount of atmospheric emissions of mercury are estimated to come from Asia (49%, of which 39% is from East and South-east Asia), followed by South America (18%), and Sub-Saharan Africa (16%). Mercury emissions from cremation, which contributed 0.17% of the total global anthropogenic mercury emissions in 2019 (UNEP 2018), are expected to increase as global cremations increase (Tibau and Grube 2019).

Table 5-6. Global Anthropogenic Emissions of Mercury to the Atmosphere by Sector

Sector	Mercury emissions (range), metric tons	Percentage of total amount
Artisanal and small-scale gold mining	838 (675–1,000)	37.7
Stationary combustion of coal from power plants	292 (255–346)	13.1
Cement production (raw materials and fuel, excluding coal)	233 (117–782)	10.5
Non-ferrous metal production (principally aluminum, copper, lead, and zinc)	228 (154–338)	10.3
Waste	147 (120–223)	6.6
Stationary combustion of coal (industrial)	126 (106–146)	5.67
Large-scale gold production	84.5 (72.3–97.4)	3.8
Vinyl-chloride monomer (mercury catalyst)	58.2 (28.0–88.8)	2.6
Stationary combustion of coal (domestic/residential, transportation)	55.8 (36.7–69.4)	2.51
Biomass burning (domestic, industrial, and power plant)	51.9 (44.3–62.1)	2.33
Pig iron and steel production (primary)	29.8 (19.1–76.0)	1.34
Chloralkali production (mercury process)	15.1 (12.2–18.3)	0.68
Waste incineration (controlled burning)	15.0 (8.9–32.3)	0.67
Oil refining	14.4 (11.5–17.2)	0.65
Mercury production	13.8 (7.9–19.7)	0.62
Secondary steel production	10.1 (7.65–18.1)	0.46
Cremation	3.77 (3.51–4.02)	0.17
Stationary combustion of oil (domestic/residential, transportation)	2.70 (2.33–3.21)	0.12
Stationary combustion of oil (power plants)	2.45 (2.17–2.84)	0.11
Stationary combustion of oil (industrial)	1.40 (1.18–1.69)	0.06
Stationary combustion of gas (power plants)	0.349 (0.285–0.435)	0.02
Stationary combustion of gas (domestic/residential, transportation)	0.165 (0.13–0.22)	0.01
Stationary combustion of gas (industrial)	0.123 (0.10–0.15)	0.01
Total	2,220 (2,000, 3,820)	100

Source: UNEP 2018

5.3.2 Water

Estimated releases of 245 pounds (~0.1 metric tons) of elemental mercury to water from 351 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024).

Estimated releases of 2,735 pounds (~1.1 metric tons) of mercury compounds to water from 1,007 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). These releases are summarized in Tables 5-4 and 5-5.

Natural weathering of mercury-bearing minerals in igneous rocks can contribute substantially to environmental mercury. An analysis conducted in 1972 estimated that this source directly released about 800 metric tons of mercury per year to surface waters of the earth (Gavis and Ferguson 1972). Atmospheric deposition of elemental mercury from both natural and anthropogenic sources has been identified as an indirect source of mercury to surface waters (WHO 1991). Mercury associated with soils can be directly washed into surface waters during rain events. Surface runoff is an important mechanism for transporting mercury from soil into surface waters, particularly for soils with high humic content (Meili 1991). Mercury may also be released to surface waters in effluents from industrial processes (Dean et al. 1972; EPA 1971; UNEP 2018).

Atmospheric deposition is a significant source of mercury emissions to water bodies. Gaseous elemental mercury can remain suspended in the atmosphere for many months and is susceptible to long-range atmospheric transport. While the United States and Canada have reduced mercury emissions significantly over the last several decades, anthropogenic emissions arising from outside of North America continue to deposit mercury into the Great Lakes Basin. The International Joint Commission (IJC) reported that Environment Canada estimates that 95% of anthropogenic deposition in Canada arises from foreign sources (IJC 2015). Cohen et al. (2016) developed a model to estimate atmospheric deposition of mercury to the Great Lakes and calculated that the United States contributed the most anthropogenic emissions (25%), followed by China (6%). Canada’s deposition of mercury to the Great Lakes in 2005 was estimated to approximately 2%, while 15% was attributed to India, Russia, and Mexico. All other nations combined were estimated to contribute a little over 4% of the total atmospheric deposition of mercury to the Great Lakes. The rest of the deposition arose from oceanic natural emissions and re-emissions of previously deposited mercury (32%), terrestrial natural emissions and re-emissions (17%), biomass burning (5.1%), and geogenic emissions such as those coming from volcanoes (6.4%).

The Mercury Deposition Network (MDN) provides long-term temporal records of total mercury deposition in precipitation across the United States and Canada. Annual deposition data from the MDN are provided in Table 5-7 for select years for the previous 2 decades.

Table 5-7. Mercury Deposition Data from the Mercury Deposition Network for North America

Year	Deposition (µg/m^2)
1998	10.56
2005	9.08
2010	8.96
2015	8.91
2018	8.78

Source: MDN 2020

According to the United Nations Global Mercury Assessment, anthropogenic global mercury emissions to waters from artisanal and small-scale gold mining was approximately 1,220 metric tons in 2015 (UNEP 2018). The majority of releases occurred in South America (53%), East and Southeast Asia (36%), and Sub-Saharan Africa (8%). Additionally, another 580 metric tons of mercury were released to water from other anthropogenic activities, primarily waste treatment, ore mining and processing, and from the energy sector.

5.3.3 Soil

Estimated releases of 36,081 pounds (~16.4 metric tons) of elemental mercury to soil from 351 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for about 62% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). An additional 145 pounds (~0.07 metric tons), accounted for <1% of the total environmental emissions, were released via underground injection (TRI22 2024). These releases are summarized in Table 5-4.

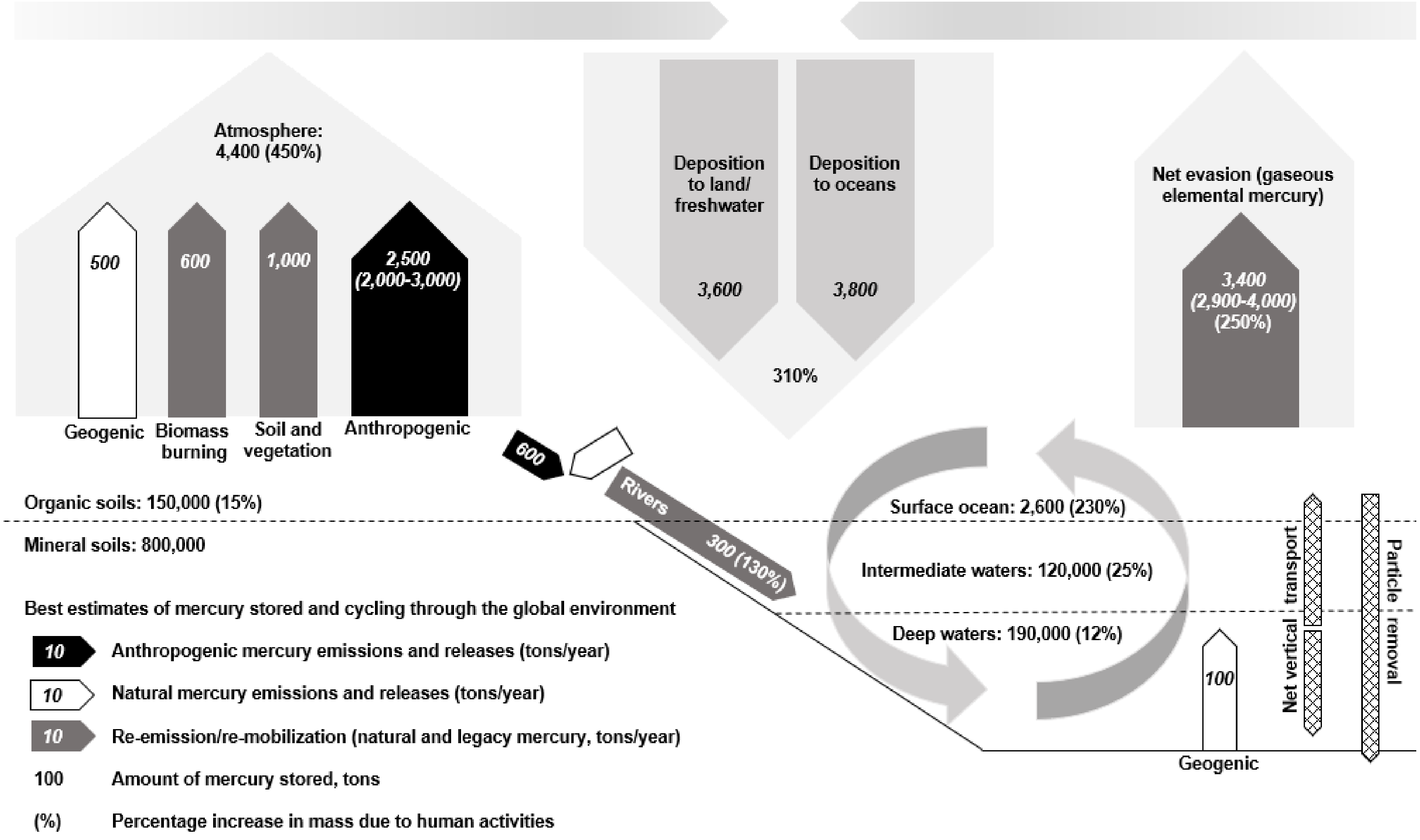
Estimated releases of 3,861,917 pounds (~1,751.7 metric tons) of mercury compounds to soil from 1,007 facilities reporting to TRI domestic manufacturing and processing facilities in 2022, accounted for about 92% of the estimated total environmental releases from facilities required to report to the TRI (TRI22 2024). An additional 315 pounds (~0.1 metric tons), accounted for <1% of the total environmental emissions, were released via underground injection (TRI22 2024). These releases are summarized in Table 5-5.

Atmospheric deposition of mercury from both natural and anthropogenic sources has been identified as an indirect source of mercury to soil and sediments (Dastoor et al. 2022; MDN 2020;). Since vapor-phase mercury is subject to long-range transport, it can be deposited to remote areas such as the Arctic. Atmospheric deposition of mercury in the Arctic was estimated as 133±9 metric tons per year in 1990 and decreased to 119±10 metric tons in 2005 (Goodsite et al. 2013). The contributions to Arctic deposition of anthropogenic mercury were dominated by sources in East Asia (32%), Commonwealth of Independent States (12%), and Africa (12%) (Dastoor et al. 2022). Risch et al. (2017) compiled data from the National Atmospheric Deposition Program for years 2007–2014 and determined that the annual deposition flux rate in deciduous-coniferous forests in 16 states in the eastern United States was approximately 11.7 $\mu g/m^2$-year (range 2.2–23.4 $\mu g/m^2$-year). Mercury is released to cultivated soils through the direct application of inorganic and organic fertilizers (e.g., sewage sludge and compost), and lime (Andersson 1979). The use of biosolids as a nutrient for agricultural soils has the potential to release mercury to soils. Lomonte et al. (2010) studied the levels of mercury in biosolids from a wastewater treatment plant in Melbourne, Australia and found concentrations between 3.5 and 8.4 mg/kg.

5.4 ENVIRONMENTAL FATE

The natural global bio-geochemical cycling of mercury is characterized by degassing of the element from soils and surface waters, followed by atmospheric transport, deposition of mercury back to land and surface waters, and sorption of the compound to soil or sediment particulates. Figure 5-3 summarizes the approximate global contributions of natural and anthropogenic sources to the mercury cycle (UNEP2018). Mercury deposited on land and open water is, in part, revolatilized back into the atmosphere. This emission, deposition, and revolatilization creates difficulties in tracing the movement of mercury to its sources (WHO 1990). Particulate-bound mercury can be converted to insoluble mercury sulfide and precipitated or bioconverted into more volatile or soluble forms that re-enter the atmosphere or are bioaccumulated in aquatic and terrestrial food chains (EPA 1984).

Figure 5-3. Global Mercury Budget: Impact of Human Activities on the Mercury Cycle and the Resulting Increase in Mercury Accumulated in Soils and Oceans



Source: UNEP (2018)

5.4.1 Transport and Partitioning

Air. Mercury has three valence states. The specific state and form of the compound found in an environmental medium is dependent upon a number of factors, including the redox potential and pH of the medium. The most reduced form is metallic or elemental mercury, which is a liquid at ambient temperatures, but readily vaporizes. Although point source emissions of mercury to the environment affects ecosystems locally, these large sources of mercury pollution can contribute to the global atmospheric level of mercury since it is a volatile substance. Deposition can occur far distances from emission sources and impact remote ecosystems (Sizmur et al. 2018). Over 95% of the mercury found in the atmosphere is gaseous mercury (Hg^0), the form involved in long-range (global) transport of the element. Residence time in the atmosphere has been estimated to range from 6 days (Andren and Nriagu 1979) to 2 years (EPA 1984). Approximately 5% of atmospheric mercury is associated with particulates, which have a shorter atmospheric residence time, are removed by dry or wet deposition, and may show a regional or local distribution pattern (Nater and Grigal 1992). Atmospheric inputs may be more significant in areas where other sources of contamination, such as contaminated rivers, are less important or nonexistent (Kelly et al. 1991). Although local sources are important, a 72-hour travel time trajectory for mercury indicates that some mercury found in rain may originate from sources up to 2,500 km (1,550 miles) away (Glass et al. 1991).

Metallic mercury released in vapor form to the atmosphere can be transported long distances before it is converted to other forms of mercury, and wet and dry deposition processes return it to land and water surfaces. Dry deposition may account for approximately 70% of the total atmospheric deposition of mercury during the summer, although on an annual basis, wet and dry deposition may be of equal importance (Lindberg et al. 1991). Up to 22% of the annual input of mercury to Lake Erie is from dry deposition of mercury-containing atmospheric particles or from precipitation (Kelly et al. 1991). Wet deposition is the primary method of removal of mercury from the atmosphere (approximately 66%) (Fitzgerald et al. 1991; Lindqvist et al. 1991) and may account for virtually all of the mercury content in remote lakes that do not receive inputs from other sources (e.g., industrial effluents) (Hurley et al. 1991; Swain et al. 1992). Most inert mercury (Hg+2) in precipitation is bound to aerosol particulates, which are relatively immobile when deposited on soil or water (Meili et al. 1991). Wet deposition has high spatial variability globally; however, a general trend of highest wet deposition in the lower and mid-latitudes of the Northern Hemisphere, with lower deposition rates in the Arctic and the Southern Hemisphere, have been observed (Lyman et al. 2020). In North America, wet deposition tends to be highest in the southeastern United States. Dry deposition flux rates of mercury have been reported as approximately

10 μg/m^2-year in Asia and approximately 6 μg/m^2-year in North America (Lyman et al. 2020). The difference is due to much higher anthropogenic emissions, and thus ambient concentrations, in Asia.

Mercury is also present in the atmosphere to a limited extent in unidentified soluble forms associated with particulate matter. In addition to wet and dry deposition processes, mercury may also be removed from the atmosphere by sorption of the vapor form to soil or water surfaces (EPA 1984). Cloud water chemical properties also affect the speciation of inorganic mercury compounds (Lyman et al. 2020). Highly acidic (pH <4) cloud water displayed 10–20 times higher total mercury concentrations than cloud water at pH >4.

Water. In soils and surface waters, mercury can exist in the mercuric (Hg^{+2}) and mercurous (Hg^{+1}) states as a number of complex ions with varying water solubilities. Mercuric mercury, present as complexes and chelates with ligands, is probably the predominant form of mercury present in surface waters. The transport and partitioning of mercury in surface waters and soils are influenced by the particular form of the compound. More than 97% of the dissolved gaseous mercury found in water consists of elemental mercury (Vandal et al. 1991). Hydrologic transport and geochemical cycling of mercury in water is an important part of the mercury global transport cycle. Wetlands, particularly estuarine marshes, are environments with biogeochemical settings that are favorable to elevated activity of the anaerobic bacteria responsible for the conversion of inorganic mercury into methylmercury (Turner et al. 2018). Conditions that favor methylmercury production in wetlands and other aquatic ecosystems include anoxic sediments, plentiful sources of labile carbon, and levels of sulfate that are neither too low to inhibit the activity of sulfate-reducing bacteria or too high, wherein significant sulfide is produced and mercury bioavailability for methylation is reduced. Turner et al. (2018) measured net fluxes of mercury from the mercury-contaminated Penobscot River and Bay system located in Maine. Over four tidal cycles on the South Marsh River, it was demonstrated that the marsh was a consistent sink over typical 12-hour tidal periods for total mercury with a flux rate of 9.2–47 μg/m^2-day and a total methylmercury flux rate of 0.2–1.4 μg/m^2-day. It was noted that the marsh was a source of methylmercury to the Penobscot River during large spring tides; however, it was concluded that Mendall Marsh is not a significant source of mercury or methylmercury to Penobscot River and Bay system.

Volatile forms (e.g., metallic mercury and dimethylmercury) are expected to evaporate to the atmosphere, whereas solid forms partition to particulates in the soil or water column and are transported downward in the water column to the sediments (Hurley et al. 1991). Vaporization of mercury from soils may be controlled by temperature, with emissions from contaminated soils being greater in warmer weather when soil microbial reduction of Hg^{+2} to the more volatile elemental mercury is greatest (Lindberg et al. 1991). Vapor-phase mercury volatilized from surface waters has been measured (Schroeder and Fanaki 1988); however, the dominant process controlling the distribution of mercury compounds in the environment appears to be the sorption of nonvolatile forms to soil and sediment particulates, with little resuspension from the sediments back into the water column (Bryan and Langston 1992). Cossa et al. (1988) found that 70% of the dissolved mercury in St. Lawrence River water was associated with organic matter. The study authors reported that the removal mechanism was flocculation of organic mercury colloids in freshwater. Methylmercury and other mercury fractions are strongly bound to organic matter in water and may be transported in runoff water from contaminated lakes to other surface waters and soils (Lee and Iverfeldt 1991). Small amounts (2–4 ng/L [ppt]) of mercury are able to move from contaminated groundwater into overlying lakes, with concentrations reaching a maximum near the sediment/water interface; however, since most of the mercury in the groundwater is derived from atmospheric sources, this low range of values indicates that most of the mercury deposited on soil (92–96% of the 10.3 $\mu g/m^2$/year of mercury deposited) is absorbed to the soil and does not leach down into the groundwater (Krabbenhoft and Babiarz 1992).

Sediment and Soil. The sorption process has been found to be related to the organic matter content of the soil or sediment. Mercury is strongly sorbed to humic materials and sesquioxides in soil at a pH >4 (Blume and Brümmer 1991) and to the surface layer of peat (Lodenius and Autio 1989). Mercury has been shown to volatilize from the surface of more acidic soils (i.e., soil pH <3.0) (Warren and Dudas 1992). Adsorption of mercury to mineral surfaces increases with increasing pH; however, increases in chloride concentration reduces the extent of adsorption (Schuster 1991). Mercury is sorbed to soil with high iron and aluminum content up to a maximum loading capacity of 15 g/kg (15,000 ppm) (Ahmad and Qureshi 1989). Inorganic mercury sorbed to particulate material is not readily desorbed. Thus, freshwater and marine sediments are important repositories for inorganic forms of the element, and leaching is a relatively insignificant transport process in soils. However, surface runoff is an important mechanism for moving mercury from soil to water, particularly for soils with high humic content (Meili 1991). Mobilization of sorbed mercury from particulates can occur through chemical or biological reduction to elemental mercury and bioconversion to volatile organic forms (Andersson 1979; EPA 1979, 1984). Metallic mercury may move through the top 3–4 cm of dry soil at atmospheric pressure; however, it is unlikely that further penetration would occur (Eichholz et al. 1988). Bachand et al. (2019) demonstrated that addition of coagulants, such as polyaluminum chloride and ferric sulfate, to soils contaminated with mercury reduced the transfer into aquatic systems by sequestering mercury into insoluble particulate forms resulting in enhanced settling of particles.

The volatilization and leaching of various forms of mercury (elemental, mercuric sulfide, mercuric oxide, and mercurous oxide) from soils or wastes were examined using the headspace method for volatilization and the Resource and Conservation Recovery Act (RCRA) leaching protocols for leaching through soil to determine if the leachates exceeded the RCRA limit of 200 μg/L (ppb) (Willett et al. 1992). With the exception of mercuric sulfide, the other forms of mercury increased in concentrations in the headspace vapor and in the leachate as the soil concentrations increased, although the elemental mercury concentrations never exceeded the RCRA limit, indicating that it was relatively nonleachable. Mercuric sulfide also did not exceed the background level for the leachate and was consistently <0.001 mg/m^3 for the vapor concentrations, indicating that it was also nonleachable and did not readily volatilize. This study also showed that concentrations of mercury in leachate could not be correlated with the concentration of mercury in the soil or in the headspace vapors (Willett et al. 1992). Mercuric sulfide has been found to strongly adsorb to soil, and even with weathering, any mercury released from the mercuric sulfide is readsorbed by the soil (Harsh and Doner 1981).

Other Media. Mercury emitted to the atmosphere is primarily in an inorganic form and it remains primarily in the inorganic form when deposition occurs to water bodies. However, in water and sediments, inorganic mercury can be transformed into methylmercury, which is very prone to bioaccumulate and biomagnify as it moves through the food chain (Riisgard and Hansen 1990; UNEP 2018). In a study of bioaccumulation of mercury in aquatic organisms, it was shown that fish had lower methylmercury bioaccumulation factors in sites with high organic carbon, especially sites with large coastal wetlands and large variability in dissolved organic carbon levels in the water column (Taylor et al. 2018b). Methylmercury in surface waters is rapidly accumulated by aquatic organisms; concentrations in carnivorous fish (e.g., pike, shark, and swordfish) at the top of both freshwater and marine food chains are biomagnified on the order of 10,000–100,000 times the concentrations found in ambient waters (EPA 1979, 1984; WHO 1990, 1991). The range in experimentally determined bioconcentration factor (BCF) values is shown in Table 5-8. The bioaccumulation potential for methylmercury in fish is influenced by the pH of the water, with a greater bioaccumulation seen in waters with lower pH (BDI 2011; Ponce and Bloom 1991). Mercury concentrations in fish have also been negatively correlated with other water quality factors, such as alkalinity and dissolved oxygen content (Ontario Ministry of the Environment 1991). Biogeochemical and ecological mechanisms are responsible for the transfer of mercury from near-shore contaminated sediments to higher trophic levels in offshore environments. Anoxic conditions in estuaries favor the production of methylmercury; however, sediments high in total organic carbon and iron sulfides tend to reduce the bioavailability of mercury in these areas. Trophic relay or bioadvection of methylmercury from sediments in contaminated estuaries to offshore pelagic organisms may occur through predator-prey relationships or the movement of juvenile species to deeper waters (Chen et al. 2009).

Table 5-8. Bioconcentration of Various Mercury Compounds by Freshwater and Saltwater Organisms

Species	Tissue	Chemical	Duration (days)	Bioconcentration factor[a]
Freshwater species				
Mercury (II)				
Rainbow trout *Salmo gairdneri*	Whole body	Mercuric chloride	60	1,800
Fathead minnow *Pimphales promelas*	Whole body	Mercuric chloride	287	4,994[b]
Organomercury compounds				
Rainbow trout *S. gairdneri*	Whole body	Methylmercuric chloride	60	11,000
Rainbow trout *S. gairdneri*	Whole body	Methylmercuric chloride	75	85,700
Brook trout *Salvelinus fontinalis*	Muscle	Methylmercuric chloride	273	11,000–33,000
Brook trout *S. fontinalis*	Whole body	Methylmercuric chloride	273	10,000–23,000
Brook trout *S. fontinalis*	Muscle and whole body	Methylmercuric chloride	756	12,000
Fathead minnow *P. promelas*	Whole body	Methylmercuric chloride	336	44,130–81,570
Saltwater species				
Mercury (II)				
Eastern oyster (adult) *Crassostrea virginica*	Soft parts	Mercuric chloride	73	10,000
American lobster (adult) *Homarus americanus*	Soft parts	Mercuric chloride	30	129
Organomercury compounds				
Eastern oyster (adult) *C. virginica*	Soft parts	Methylmercuric chloride	74	40,000
Eastern oyster (adult) *C. virginica*	Soft parts	Phenylmercuric chloride	74	40,000

[a] Results are based on the concentration of mercury, not the concentration of the mercury compound to which the animal was exposed.
[b] From concentrations that caused adverse effects in a life-cycle test.

Source: ASTER 1997

Mercury levels in freshwater fish have been shown to be elevated in areas impacted by gold mining operations (Diringer et al. 2015; Salazar-Camacho et al. 2021). A study of fish impacted by gold mining in the Atrato River Basin, Columbia identified a correlation between total mercury levels with fish length and trophic level, indicating that mercury was biomagnifying in the ecosystem. The median total mercury level in all carnivorous fish (n=533) was 225.4±344.3 μg/kg and the median level in all non-carnivorous fish (n=289) was 82.4±91.3 μg/kg. Total mercury concentrations (median±SD) in the fish categorized by trophic level are shown in Table 5-9.

Table 5-9. Mercury Concentrations Found in Fish

Species	Total mercury median (±SD) μg/kg	Trophic level
Doncella *Ageneiosus pardalis*	678.5±344.9	3.8
Moncholo *Hoplias malabaricus*	401.4±278.5	4.5
Caga *Trachelyopterus fisheri*	374.3±250.0	3.5
Bagre sapo *Pseudopimelodus schultzi*	432.7±897.0	3.7
Agujeta *Ctenolucius beani*	270.9±289.1	4
Majarra Negra *Caquetaia umbrifera*	218.5±221.1	3.8
Mayupa *Sternopygus macrurus*	177.0±452.2	3.2
Mojarra Amarilla *Caquetaia kraussii*	218.0±200.6	3.4
Liso *Rhamdia quelen*	145.8±211.6	3.9
Sardina colirroja *Astyanax fasciatus*	117.2±113.7	3
Charre *Pimelodus punctatus*	100.5±66.9	3.3
Cocobolo *Andinoacara pulcher*	116.7±59.7	3.3
Dentón *Leporinus muyscorum*	116.7±68.0	2.2
Bocachico *Prochilodus magdalenae*	93.1±113.2	2.1
Guacuco *Hypostomus hondae*	56.0±55.5	2
Viejita *Cyphocharax magdalenae*	32.0±53.2	2

Source: Salazar-Camacho et al. 2021

Ruus et al. (2015) analyzed different species in the Kongsfjorden food web system in Norway and found that tissue concentrations of methylmercury increased with increasing trophic level and were highly correlated with total mercury levels. The highest levels of total mercury and methylmercury were observed in birds (95–1,108 ng/g), which were at the highest trophic level in the food web, and the lowest levels were found in zooplankton (4.7–7.5 ng/g), the lowest trophic level in the foodweb.

Biomagnification factors for methylmercury in the food webs of Lake Ontario were lowest for the transfer of methylmercury from mysids to amphipods (1.1), plankton to amphipods (1.8), and plankton to mysids (2.4); intermediate for the transfer from mysids to fish (5.1) and amphipods to fish (6.5); and highest for the transfer from plankton to fish (10.4) (Evans et al. 1991). The biomagnification of methylmercury from water through several trophic levels is compared to the biomagnification of inorganic mercury in Table 5-10. Watras and Bloom (1992) reported that biomagnification of methylmercury in Little Rock Lake seems to be the result of two processes: the higher affinity of inorganic mercury in lower trophic level organisms and the high affinity of methylmercury in fish. Fish appear to accumulate methylmercury from both food sources and the water column. However, Hall et al. (1997) found that food was the predominant source of mercury uptake in fish. The BCF of methylmercury in fish in Little Rock Lake was 3×10^6 (Porcella 1994). Mason et al. (1995) also compared bioaccumulation of inorganic mercury and methylmercury and found that passive uptake of the mercury complexes ($HgCl_2$ and CH_3HgCl) results in high concentrations of both the inorganic and methylated mercury in phytoplankton. However, differences in partitioning within phytoplankton cells between inorganic mercury (which is principally membrane-bound) and methylmercury (which accumulated in the cytoplasm) led to a greater assimilation of methylmercury during zooplankton grazing.

Table 5-10. Comparison of the Biomagnification of Methylmercury and Inorganic Mercury in a Freshwater Food Chain (Little Rock Lake)

Medium or trophic level	Methylmercury	Inorganic mercury	Percent methylmercury
Water	1	10	10
Phytoplankton	10^5	$10^{5.7}$	15
Zooplankton	$10^{5.5}$	$10^{5.9}$	30
Fish	$10^{6.5}$	10^5	95

Source: Watras and Bloom 1992

Rumbold et al. (2018) studied biomagnification of mercury in two different locations along the Florida Keys using 50 different species of fish to determine whether the high biodiversity in coral reefs in these complex food webs reduces the biomagnification potential in these regions. It was determined that the Trophic Magnification Slopes (TMS) were very similar for the two sites and that these TMS were also within the ranges of slopes reported for food webs in other ecosystems; therefore, biomagnification of mercury in fish muscle tissue was not reduced in this complex ecosystem. In a similar study conducted along the coast of the Florida Keys, Thera and Rumbold (2014) assessed the trophic biomagnification factor of mercury using 57 species of invertebrates. They concluded that the levels of mercury increased by a factor of 5 with each unit increase in trophic level.

Most of the discrimination between inorganic and methylmercury thus occurs during trophic transfer, while the major enrichment factor is between water and the phytoplankton. This also has been reported for the diatom, *Thalassiosura weissflogii*, in a marine food chain (Mason et al. 1996). Methylmercury was accumulated in the cell cytoplasm, and its assimilation by copepods was 4 times more efficient than the assimilation of inorganic mercury. Bioaccumulation has been demonstrated for predator fish in both freshwater and marine systems and in marine mammals.

Aquatic macrophytes have been found to bioconcentrate methylmercury in almost direct proportion to the mercury concentration in the water (Ribeyre et al. 1991). Mortimer (1985) reported BCFs for several species of submerged aquatic plants exposed to inorganic mercury in laboratory aquaria of 3,300, 1.3, 0.9, and 1.3 for *Utricularia*, *Ceratophyllum*, *Najas*, and *Nitella*, respectively. The concentration factors used by this author was based on µg g^{-1} dry weight in the plant/µg mL^{-1} water day^{-1}.

The potential for bioaccumulation in terrestrial food chains is demonstrated by the uptake of mercury by the edible mushroom, *Pleurotus ostreatus*, grown on compost and containing mercury at concentrations of up to 0.2 mg/kg (ppm). The bioaccumulation factors reported ranged from 65 to 140, indicating that there are potential risks to human health if these mushrooms are eaten in large quantities (Bressa et al. 1988). Elevated concentrations of mercury in 149 samples of mushrooms representing 11 different species were reported by Kalač et al. (1991). The study authors collected mushrooms within 6 km of a lead smelter in Czechoslovakia in operation since 1786. Mercury was accumulated by *Lepista nuda* and *Lepiota rhacodes* at 11.9 mg/kg (ppm) and 6.5 mg/kg (ppm) (dry weight), respectively. The mean concentration of other species ranged from 0.3 to 2.4 mg/kg (ppm). Concentrations of mercury in most of the mushroom species collected in that location were higher than in mushrooms collected in other parts of the country.

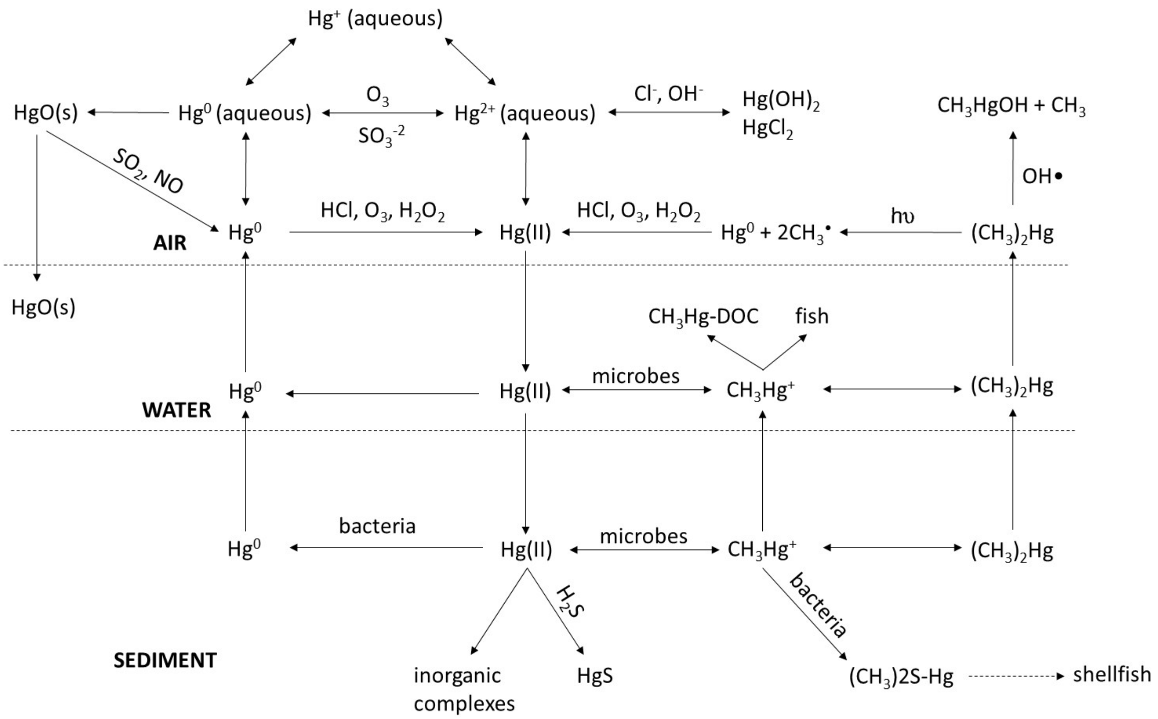
Data from higher plants indicate that virtually no mercury is taken up from the soil into the shoots of plants such as peas, although mercury concentrations in the roots may be significantly elevated and reflect the mercury concentrations of the surrounding soil (Lindqvist et al. 1991). In a study by Granato et al. (1995), municipal solid waste sludge mercury concentrations from the Metropolitan Water Reclamation District of Greater Chicago were found to range from 1.1 to 8.5 mg/kg (ppm), with a mean concentration of 3.3 mg/kg (ppm). From 1971 to 1995, sludge applications were made to a Fulton County, Illinois sludge utilization site. About 80–100% of the mercury applied to the soils in sewage sludge since 1971 still resided in the top 15 cm of soil. The study authors reported that sewage sludge applications did not increase plant tissue mercury concentrations in corn or wheat raised on the sludge utilization site.

Earthworms, *Lumbricus sp.*, bioaccumulate mercury under laboratory and field conditions in amounts that are dependent on soil concentrations and exposure duration (Cocking et al. 1994). Maximum mercury tissue concentrations in laboratory cultures were only 20% of the 10–14.8 μg/g (ppm) (dry weight) observed in individual worms collected from contaminated soils (21 μg/g) on the South River flood plain at Waynesboro, Virginia. Bioconcentration occurred under field conditions in uncontaminated control soil (0.2 μg Hg/g); however, total tissue mercury concentrations (0.4–0.8 μg/g dry weight) were only 1–5% of those for earthworms collected on contaminated soils. Uptake by the earthworms appeared to be enhanced in slightly acidic soils (pH 5.9–6.0) in laboratory cultures. Soil and earthworm tissue mercury contents were positively correlated under both field and laboratory conditions. Predation of earthworms contaminated with mercury could pass the contamination to such predators as moles and ground-feeding birds, such as robins (Cocking et al. 1994).

5.4.2 Transformation and Degradation

Mercury is transformed in the environment by biotic and abiotic oxidation and reduction, bioconversion of inorganic and organic forms, and photolysis of organomercurials. Inorganic mercury can be methylated by microorganisms indigenous to soils, fresh water, and salt water (Blanc et al. 2018). This process is mediated by various microbial populations under both aerobic and anaerobic conditions. The most probable mechanism for this reaction involves the nonenzymatic methylation of mercuric mercury ions by methyl cobalamin compounds produced as a result of bacterial synthesis. Mercury forms stable complexes with organic compounds. Monoalkyl mercury compounds (e.g., methylmercuric chloride) are relatively soluble; however, the solubility of methylmercury is decreased with increasing dissolved organic carbon content, indicating that it is bound by organic matter in water (Miskimmin 1991). Dialkyl mercury compounds (e.g., dimethyl-mercury) are relatively insoluble (EPA 1979, 1984). Dimethylmercury is volatile, although it makes up <3% of the dissolved gaseous mercury found in water (Andersson et al. 1990; Vandal et al. 1991). The major pathways for transformation of mercury and various mercury compounds in air, water, and soil are shown in Figure 5-4.

Figure 5-4. Transformation of Mercury in Air, Water, and Sediment



Dashed lines represent the boundary between environmental compartments.

Air. Lyman et al. (2020) published an updated review of the atmospheric transport, partitioning, and transformation of mercury. Oxidation by ozone and hydroxyl radicals has historically been assumed to be the dominant oxidation mechanism for ambient elemental mercury; however, recent evidence suggests that oxidation by bromine radicals may be a globally important oxidation mechanism. Ye et al. (2016), showed that in the marine boundary layer, bromine and bromine oxide were the dominant gaseous elemental mercury oxidants, with mixing ratios reaching 0.1 and 1 pptv, respectively, and contributing approximately 70% of the total gaseous oxidized mercury production during midday, while ozone dominated oxidation (50–90 % of gaseous oxidized mercury production) over the remaining day when bromine and bromine oxide mixing ratios decreased.

The overall residence time of elemental mercury in the atmosphere has been estimated to be 6 days to 2 years, although in clouds, a fast oxidation reaction on the order of hours may occur between elemental mercury and ozone. Some mercury compounds, such as mercuric sulfide, are quite stable in the atmosphere as a result of their binding to particles in the aerosol phase (Lindqvist et al. 1991). Other mercury compounds, such as mercuric hydroxide ($Hg[OH]^2$), which may be found in the aqueous phase of the atmosphere (e.g., rain), are rapidly reduced to monovalent mercury in sunlight (Munthe and McElroy 1992). The main atmospheric transformation process for organomercurials appears to be photolysis (EPA 1984; Johnson and Braman 1974; Williston 1968).

Water. The most important transformation process in the environmental fate of mercury in surface waters is the methylation and demethylation cycle. The methylation of mercury in surface waters is largely driven by the presence of sulphate-reducing bacteria, and demethylation can occur through both abiotic and biotic means (Ouddane et al. 2015; Turner et al. 2018). The photochemical degradation of methylmercury is the most important process of the demethylation cycle at the surface or in shallow waters. Demethylation by biotic oxidation and reduction reactions are the dominant pathways at deeper depths and in sediments.

Any form of mercury entering surface waters can be microbially converted to methylmercury, given favorable conditions. The major factors that influence the rate of methylation are the abundance of anaerobic microorganisms that have the biochemistry to methylate mercury and the bioavailability of inorganic Hg(II) in these microorganisms (SERDP 2014). Sulfur-reducing bacteria are responsible for most of the mercury methylation in the environment (Gilmour and Henry 1991; Ouddane et al. 2015; Turner et al. 2018), with anaerobic conditions favoring their activity (Regnell and Tunlid 1991). Yeasts, such as *Candida albicans* and *Saccharomyces cerevisiae*, whose growth is favored by low pH conditions, are able to methylate mercury and are also able to reduce ionic mercury to elemental mercury (Yannai et al. 1991). Methyl cobalamin compounds produced by bacterial synthesis appear to be involved in the nonenzymatic methylation of inorganic mercury ions (Regnell and Tunlid 1991). The rate of methylmercury formation by this process is largely determined by the concentration of methyl cobalamin compounds, inorganic mercuric ions, and the oxygen concentration of the water, with the rate increasing as the conditions become anaerobic. Volatile elemental mercury may be formed through the demethylation of methylmercury or the reduction of inorganic mercury, with anaerobic conditions again favoring the demethylation of the methylmercury (Barkay et al. 1989; EPA 1979; Regnell and Tunlid 1991). Increased dissolved organic carbon levels reduce methylation of mercury in the water column (Gilmour and Henry 1991), possibly as a result of the binding of free mercury ions to the dissolved organic carbon at low pH, thus reducing their availability for methylation, or the dissolved organic carbon may inhibit the methylating bacteria (Miskimmin et al. 1992). Alternatively, low pH favors the methylation of mercury in the water column, particularly in acid deposition lakes, while inhibiting its demethylation (Gilmour and Henry 1991). It has also been shown that the methylation rate is not affected by addition of sulfate in softwater lakes (Kerry et al. 1991). Even in polar marine systems, methylation of mercury can occur. Sea water and ice samples were collected in the East Antarctic Sea and analyzed for total mercury and methylmercury (Gionfriddo et al. 2016). The marine nitrite-oxidizing bacterium, Nitrospina, was identified as the most likely source for microbially induced methylation of mercury in polar ice and water.

At a pH of 4–9 and a normal sulfide concentration, mercury will form mercuric sulfide. This compound is relatively insoluble in aqueous solution ($11\text{x}10^{-17}$ ppb), and it will therefore precipitate out and remove mercury ions from the water, reducing the availability of mercury to fish. Under acidic conditions, however, the activity of the sulfide ion decreases, thus inhibiting the formation of mercuric sulfide and favoring the formation of methylmercury (Bjornberg et al. 1988). Low pH and high mercury sediment concentrations favor the formation of methylmercury, which has greater bioavailability potential for aquatic organisms than inorganic mercury compounds. Methylmercury may be ingested by aquatic organisms lower in the food chain, such as yellow perch, which in turn are consumed by piscivorous fish higher in the food chain (Cope et al. 1990; Wiener et al. 1990). Mercury cycling occurs in freshwater lakes, with the concentrations and speciation of the mercury being dependent on limnological features and water stratification. Surface waters may be saturated with volatile elemental mercury, whereas sediments are the primary source of the mercury in surface waters. During the summer months, surface concentrations of methyl and elemental mercury decline as a result of evaporation, although they remain relatively constant in deeper waters (Bloom and Effler 1990).

Abiotic reduction of inorganic mercury to metallic mercury in aqueous systems can also occur, particularly in the presence of soluble humic substances (i.e., acidic waters containing humic and fulvic acids). This reduction process is enhanced by light, occurs under both aerobic and anaerobic conditions, and is inhibited by competition from chloride ions (Allard and Arsenie 1991).

Sediment and Soil. Mercury compounds in soils may undergo the same chemical and biological transformations described for surface waters. Mercuric mercury usually forms various complexes with chloride and hydroxide ions in soils; the specific complexes formed depend on the pH, salt content, and composition of the soil solution. Formation and degradation of organic mercurials in soils appear to be mediated by the same types of microbial processes occurring in surface waters and may also occur through abiotic processes (Andersson 1979). Elevated levels of chloride ions reduce methylation of mercury in river sediments, sludge, and soil (Olson et al. 1991), although increased levels of organic carbon and sulfate ions increase methylation in sediments (Gilmour and Henry 1991). In freshwater and estuarine ecosystems, the presence of chloride ions (0.02 M) may accelerate the release of mercury from sediments (Wang et al. 1991). Cesario et al. (2017) studied the methylation and demethylation kinetics of mercury in sediments, with and without salt-marsh plant vegetation, obtained from the Guadiana and Tagus Estuaries, Portugal. Methylation and demethylation rates varied between sediments of the estuaries depending upon the presence of vegetation and the macrophyte species present. The highest methylation rate constant was observed in *Sarcocornia fruticosa* vegetated sediments at the Castro Marim site in Guadiana (Km = 0.160 day^{-1},) and the lowest rate constant was observed in non-vegetated sediments at the Alcochete site in Tagus (Km = 0.009 day^{-1}). The role that vegetation plays in methylation cycling of mercury was studied by examining the seasonal variation of methylmercury production in three types of flooded agricultural wetlands (white rice, wild rice, and fallow fields), and two permanently flooded, non-agricultural managed wetland areas in the Yolo Bypass Wildlife Unit California (Windham-Myers et al. 2014). In addition to transport and transformation reactions that occur in air, water, soil and sediment, plant biomass also plays an indirect role in the methylation and demethylation cycle of mercury. Inorganic mercury may be taken up through plant roots in contaminated soils or deposited on leafy surfaces following atmospheric deposition, whereas methylmercury appears to be primarily taken up by plants via the root system (Windham-Myers et al. 2014). Thus, the transport of methylmercury into plants in wetland environments provides a temporary storage sink and reduces the levels in the surrounding aquatic environments. However, decomposition of plant tissue, deforestation, clearing of wetlands, and fires can release stored methylmercury to other environmental media. Methylmercury levels in rice leaves tended to increase from summer to fall during the growing season and were shown to correlate with root methylmercury levels. Degrading litter residue was shown to correlate with increased methylmercury production during the winter months in the wetlands.

The rates of mercury methylation and demethylation were studied in sediments obtained from the Deûle River, France (Ouddane et al. 2015). Sampling was conducted at two locations that were contaminated with mercury from previous industrial activity. At the first location, demethylation, rather than methylation, was the dominant process. The study authors surmised that high levels of mercury found in this heavily contaminated site resulted in an increase in the rate of bacterial demethylation, either by reduction producing Hg^0 or oxidation yielding Hg^{2+} and CO_2. For the second site sampled, where total mercury levels were lower than the first site, the average net methylation potential was positive (methylation occurred at a greater rate than demethylation) even though there were sediments obtained at some depths where the rate of demethylation was greater than methylation.

In the late 1950s, unknown quantities of mercuric nitrate and elemental mercury were released into East Fork Poplar Creek from a government facility in Oak Ridge, Tennessee. Total mercury concentrations in the flood plain soil along the creek ranged from 0.5 to 3,000 ppm (Revis et al. 1989). The form of that mercury has been reported to be primarily mercuric sulfide (85–88%), with 6–9% present as elemental mercury (Revis et al. 1989, 1990). A very small amount was detected in the form of methylmercury (<0.02%). The reported presence of the mercuric sulfide suggests that the predominant biological reaction in soil for mercury is the reduction of Hg^{+2} to mercuric sulfide by sulfate-reducing bacteria under anaerobic conditions (Revis et al. 1989, 1990). Mercuric sulfide has very limited water solubility (4.5×10^{-24} mol/L), and thus, in the absence of other solvents, is likely to have limited mobility in soil. Aerobic microorganisms can solubilize Hg^{+2} from mercuric sulfide by oxidizing the sulfide through sulfite to sulfate, with the Hg^{+2} being reduced to elemental mercury (Wood 1974). However, examination of the weathering of mercuric sulfide indicated that mercuric sulfide does not undergo significant weathering when bound to riverwash soil with a pH of 6.8, although degradation may be increased in the presence of chloride and iron (Harsh and Doner 1981).

Mercury, frequently present in mine tailings, was toxic to bacteria isolated from a marsh treatment system used to treat municipal wastewaters. The minimum concentration that inhibited the bacteria (as determined by intracellular ATP levels) was approximately 0.07±0.15 mg/L (ppm) (Desjardins et al. 1988).

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to mercury depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of mercury 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 mercury 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 5-11 shows the lowest limit of detections that are achieved by analytical analysis in environmental media. An overview summary of the ranges of concentrations detected worldwide in environmental media is presented in Table 5-12.

Table 5-11. Lowest Limit of Detection Based on Standards[a,b]

Media	Detection limit	Reference
Air	30 pg/m^3 (particulate); 45 pg/m^3 (vapor)	EPA 1999a (Method IO-5)
Drinking water	0.2 ng/L	EPA 2002a (Method 1631E)
Surface water and groundwater	0.2 ng/L	EPA 2002a (Method 1631E)
Soil	4.8 μg/kg	Frentiu et al. 2013
Sediment	4.8 μg/kg	Frentiu et al. 2013
Whole blood	0.2–0.33 μg/L	CDC 2024

[a] Detection limits based on using appropriate preparation and analytics. These limits may not be possible in all situations.
[b] A review of analytical methods to detect mercury in environmental matrices has been published by Suvarapu and Baek (2017).

Table 5-12. Summary of Environmental Levels of Mercury Worldwide

Media	Low	High	For more information
Outdoor air (ng/m^3)	0.000161 (particulate)	174,000 (vapor)	Section 5.5.1
Indoor air (ng/m^3)	<3	1,500	Section 5.5.1
Surface water (ppb)	<0.002	0.09	Section 5.5.2
Ground water (ppb)	0.21	300	Section 5.5.2
Drinking water (ppb)	<0.025	300	Section 5.5.2
Food (ppm)	<0.01	0.509	Section 5.5.4
Soil (ppm)	0.063	141,000	Section 5.5.3

Detections of mercury in air, water, and soil at NPL sites are summarized in Table 5-13.

Table 5-13. Mercury Levels in Water, Soil, and Air at National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
Water (ppb)	2	3.50	16.3	316	188
Soil (ppb)	1,800	2,730	28.2	381	220

Table 5-13. Mercury Levels in Water, Soil, and Air at National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
Air (ppbv)	0.301	0.282	75.4	26	18

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

The vast majority of mercury found in the atmosphere occurs in the form of Hg^0 (gaseous elemental mercury). Total gaseous mercury (TGM) represents the sum of all gaseous compounds and gaseous elemental mercury and has been reported to have a global background concentration typically in the range of 1.5–2.0 ng/m^3, although levels can vary due to local sources (Gworek et al. 2017). Decreases in gaseous elemental mercury levels have been reported globally as mercury emissions have declined in many parts of the world. Annual decreasing trends of 1.4 and 2.7% per year in the Northern and Southern Hemispheres from 1996 to 2009, respectively, have been reported (Lyman et al. 2020). The Global Mercury Observation System (GMOS) is a European Union project consisting of 43 globally distributed monitoring stations that measure ambient atmospheric mercury levels on a global scale (Sprovieri et al. 2016). Results from GMOS data for 2013 and 2014 had reported mean background levels of 1.22 and 1.23 ng/m^3, respectively, in tropical zones and 0.93 and 0.97 ng/m^3, respectively, in the Southern Hemisphere. The 2013 and 2014 annual mean mercury concentrations were reported as 1.55 and 1.51 ng/m^3, respectively, in the Northern Hemisphere (Sprovieri et al. 2016).

Data on mercury air levels are available at the Air Quality System (AQS) database, which contains ambient air quality data collected by EPA, state, local, and tribal air pollution control agencies from monitors throughout the country. For 2019, the arithmetic mean vapor phase levels ranged from 1.11 to 2.22 ng/m^3, with a maximum level of 248 ng/m^3 reported in Milwaukee, Wisconsin (EPA 2019). The arithmetic mean mercury concentration of total suspended particulates (TSP) ranged from 0.009 to 0.0025 ng/m^3. The arithmetic mean mercury levels associated with PM_{10} (particulate matter ≤10-micron diameters) ranged from 0.00619 to 1.50 ng/m^3, while the arithmetic means for mercury associated with $PM_{2.5}$ were 0.000161–0.000317 ng/m^3. For 2023, the arithmetic mean TSP ranged from 0.00032 to 2 ng/m^3 (EPA 2024).

The Atmospheric Mercury Network (AMN) and the National Atmospheric Deposition Program (NADP) operate monitoring sites to record temporal atmospheric concentrations of mercury and deposition rates in North America. The average gaseous elemental mercury concentrations from the AMN from 2012 to 2022 are shown in Table 5-14.

Table 5-14. Atmospheric Mercury Network (AMN) Average Gaseous Elemental Mercury Concentrations, 2012–2022

Year	Averaged gaseous elemental mercury level (ng/m^3)
2012	1.45
2013	1.46
2014	1.47
2015	1.42
2016	1.44
2017	1.80
2018	1.73
2019	1.61
2020	1.76
2021	1.72
2022	1.40

Source: NADP 2024

As mercury is subject to long-range transport, it has been detected in Arctic atmospheric samples. The median (±SD) concentrations of mercury at Alert, Canada in the fall, winter, spring, and summer were 1.49 (±0.11), 1.59 (±0.17), 1.24 (±0.53), and 1.80 (±0.35) ng/m^3, respectively (Kirk et al. 2012). Atmospheric mercury trends in the Arctic from 1995 to 2018 have shown seasonal variability and generally decreasing levels after 2010 (MacSween et al. 2022). For example, over the sampling period the location at Villum Research Station, Greenland showed the largest decrease in total gaseous mercury, averaging about a 4.55 % decline per year during winter months.

Several decades ago, ambient atmospheric levels of mercury could have been an order of magnitude higher than current expected background levels. In 1990, metallic mercury concentrations in the gas and aerosol phases of the atmosphere in Sweden were 2–6 and 0.01–0.1 ng/m^3, respectively (Brosset and Lord 1991). Higher levels (10–15 μg/m^3) have been detected near point emission sources, such as mercury mines, refineries, and agricultural fields treated with mercury fungicides. Atmospheric concentrations of mercury over lakes in Wisconsin averaged 2.0 ng/m^3 (Wiener et al. 1990). Mercury levels ranged from

6.3 to 16.0 ng/m^3 above the water surface of the mercury-contaminated Wabigoon River in Ontario (Schroeder and Fanaki 1988). The mean vapor concentration of mercury in air over a forested watershed (Walker Branch Watershed) in Tennessee was 5.5 ng/m^3 in 1988–1989, while the particle-associated aerosol mercury concentration was determined to be 0.03 ng/m^3, or approximately 0.5% of the total atmospheric mercury (Lindberg et al. 1991). Lindberg et al. (1994) measured mercury vapor at concentrations of 2–6 ng/m^3 and particulate mercury at 0.002–0.06 ng/m^3 at Walker Branch Watershed, Tennessee, from August 1991 to April 1992. Particulate mercury concentrations are greater in precipitation than in ambient air. In the St. Louis River estuary, mercury levels in precipitation averaged 22 ng/L (ppt), although ambient air levels averaged 3 ng/m^3 (Glass et al. 1990).

Total gaseous mercury was measured (1992–1993) as part of the Florida Atmospheric Mercury Study (FAMS) (Gill et al. 1995). Average total gaseous mercury concentrations for 3–6-day integrated samples ranged from 1.43 to 3.11 ng/m^3 (mean 1.64 ng/m^3). In the same study, Dvonch et al. (1995) reported that the mean concentrations of total gaseous mercury measured at two inland Florida sites were significantly higher (3.3 and 2.8 ng/m^3) than measurements at an Atlantic coastal site (1.8 ng/m^3). The mean concentrations of particle-phase mercury collected at the inland sites (51 and 49 pg/m^3) were 50% higher than those at the coastal site (34 pg/m^3). The mean mercury concentration in rain samples was 44 ng/L (ppt) (range 14–130 ng/L). Guentzel et al. (1995) also reported results of the FAMS from 1992 to 1994. Particle-phase measurements ranged from 2 to 18 pg/m^3 at all sites. Measurements of monomethylmercury in precipitation ranged from <0.005 to 0.020 ng/L (ppt).

Keeler et al. (1994) measured atmospheric mercury in the Great Lakes Basin. The study authors reported that vapor-phase mercury levels were 4 times higher in Chicago, Illinois, than in South Haven, Michigan (8.7 versus 2.0 ng/m^3). Furthermore, a diurnal pattern was observed in the vapor-phase mercury levels measured at the Chicago site. The average concentration (ng/m^3) was 3.3 times greater for the daytime samples (8 AM to 2 PM) than for the night samples (8 PM to 8 AM), and the average concentration for the afternoon samples (2 PM to 8 PM) was 2.1 times greater than the night samples (average, 3.7 ng/m^3). Particulate-phase mercury concentrations were also higher at the Chicago site than at the South Haven site (98 versus 19 pg/m^3). Burke et al. (1995) reported that the concentration of mercury in vapor-phase samples measured over Lake Champlain was consistent with other rural areas (mean 2.0 ng/m^3; range 1.2–4.2 ng/m^3), and the concentrations were consistent across all seasons. Particulate-phase mercury concentrations averaged 11 pg/m^3, with the highest concentrations detected during the winter.

In the past, mercury was a component of indoor and outdoor, water-based latex paints; however, in 1990, most uses of mercury for interior and exterior paints and coatings were voluntarily cancelled by the manufacturers (DeVito and Brooks 2013). Prior to this cancellation, mercury levels in homes and buildings that were recently painted could be much higher than background levels. Indoor air mercury concentrations were determined in 37 houses in Ohio that had been painted with latex paint (Beusterien et al. 1991). Of the 37 homes studied, 21 homes had been painted with interior latex paint containing mercury a median of 86 days earlier, while the 16 control homes had not been recently painted with mercury-containing latex paints. Paint samples from the exposed homes contained a median concentration of 210 mg/L mercury (ppm) (range 120–610 mg/L). The median air mercury concentration (0.3 $\mu g/m^3$) was found to be significantly higher ($p<0.0001$) in the exposed homes (range: not detectable to 1.5 $\mu g/m^3$) than in the unexposed homes (range: not detectable to 0.3 $\mu g/m^3$). Among the exposed homes, there were seven in which paint containing <200 mg/L mercury had been applied. In these homes, the median air mercury concentration was 0.2 $\mu g/m^3$ (range: not detectable to 1 $\mu g/m^3$). Six exposed homes had air mercury concentrations >0.5 $\mu g/m^3$. The study authors reported that elemental mercury was the form of mercury released to the air and that potentially hazardous mercury exposure could occur in homes recently painted with paint containing <200 mg Hg/L (Beusterien et al. 1991). In an indoor exposure study of families of workers at a chloralkali plant in Charleston, Tennessee, mercury levels in the air of the workers' homes averaged 0.92 $\mu g/m^3$ (ATSDR 1990).

A monitoring program established at a facility at Oak Ridge National Laboratories found that the major sources of mercury release to the air were vaporization from soil, burning of coal for a steam plant, and fugitive exhaust from a former lithium isotope separation facility contaminated with mercury (DOI 1991). When the monitoring program began in 1986, ambient air mercury vapor concentrations at the facility ranged from 0.011 to 0.108 $\mu g/m^3$. These values decreased to 0.006–0.071 $\mu g/m^3$ by 1990, while background levels near the facility remained at 0.006 $\mu g/m^3$. The decrease in mercury vapor concentrations occurred primarily as a result of an 80% reduction in coal burning at the steam plant; however, periods of drought and activities such as moving contaminated soil for construction were found to increase the atmospheric mercury concentrations on a transient basis (DOI 1991). Turner and Bogle (1993) monitored ambient air for mercury around the same industrial complex site at Oak Ridge, Tennessee. Elemental mercury was used in large quantities at the nuclear weapons plant between 1950 and 1963 in a process similar to chloralkali production. Soil and water contamination had been found at the site. The results of weekly ambient monitoring for gaseous mercury from 1986 through 1990 showed that gaseous mercury levels were well below the National Emission Standard for Hazardous Air Pollutants (1.0 mg/m^3) with the exception of one station. Mean mercury levels at the control site ranged from 5 to 6 µg/m^3, while levels at the on-site stations ranged from 6–11, 11–143, 68–174, 71–109, and 4–46 µg/m^3, depending on the station. Mean particulate mercury levels were 0.00003 µg/m^3 at the control site, compared with mean concentrations at the on-site stations ranging from 0.00006 to 0.00024 µg/m^3 (Turner and Bogle 1993).

5.5.2 Water

The EPA maintains a Water Quality Portal (WQP) database that aggregates environmental monitoring data from the National Water Information System (NWIS) and STORage and RETrieval (STORET) system, providing ample data for assessment of current conditions and trends in mercury concentrations in waters of the United States. A summary of the data for ambient surface and groundwater from recent years is reported in Table 5-15 (WQP 2024).

Table 5-15. Summary of Concentrations of Dissolved Total Mercury (µg/L) Measured in Surface Water and Groundwater Across the United States

Year	Average	Maximum	Number of samples	Percent detected
Surface water				
2018	0.005	0.82	2358	44%
2019	0.030	2.59	2155	44%
2020	0.003	0.26	1821	54%
2021	0.020	4.9	1600	59%
2022	0.090	23.2	1384	54%
2023	0.044	21.0	1235	52%
Groundwater				
2018	0.11	3.3	495	29%
2019	0.33	3.4	375	23%
2020	0.12	0.86	288	11%
2021	0.09	0.69	337	15%
2022	0.17	0.95	238	5%
2023	0.32	2.39	247	8%

Source: WQP 2024

Rainwater collected in monitoring stations in the U.S Great Lakes region (2002–2008) found annual average mean concentrations at 37 sites to range from 5.6 to 13.6 ng/L (ppt) and temporal trends for decreasing and increasing concentrations, depending on location (Risch et al. 2012). Average total mercury and methylmercury levels in cloud water above the Pacific Ocean near the coast of California were 9.2±6.0 and 0.87 ±0.66 ng/L, respectively, during a sampling period from the summer of 2016

(Weiss-Penzias et al. 2018). Fitzgerald et al. (1991) measured total mercury in rainwater from May through August 1989 at Little Rock Lake, Wisconsin. The total mercury concentrations ranged from 3.2 to 15.2 ng/L (ppt). Mercury concentrations in precipitation collected in Minnesota during 1988 and 1989 averaged 18 ng/L (ppt) for an average annual mercury deposition of 15 μg/m^2 (Glass et al. 1991). Antarctic surface snow contained a mean mercury concentration of <1 pg/g (ppt) (Dick et al. 1990). Snowpack from the Muskeg watershed in Alberta, Canada contained total mercury and methylmercury levels of 0.5–38.2 and 0.01–0.27 ng/L, respectively (Wasiuta et al. 2019). In Ontario, Canada, mercury present in precipitation at an average concentration of 10 ng/L (ppt) accounted for more than half of the mercury inputs to surface waters compared with inputs from stream runoff, suggesting that atmospheric deposition is a significant source of mercury in surface waters (Mierle 1990). Lindberg et al. (1994) measured total mercury in rain collected at Walker Branch Watershed, Tennessee from August 1991 to April 1992. Rain concentrations of total mercury ranged from 7.57 ng/L (ppt) in February 1992 to 17.4 ng/L (ppt) in April 1992. Burke et al. (1995) reported that the average concentration of mercury in precipitation samples measured over Lake Champlain was 8.3 ng/L (ppt) for the sampling year, and the average amount of mercury deposited per precipitation event was 0.069 μg/m^2. The highest concentrations of mercury in precipitation samples occurred during spring and summer months. Guentzel et al. (1995) reported results of the Florida Atmospheric Monitoring Study from 1992 to 1994. The study authors found that the wet season in south Florida accounted for 80–90% of the annual rainfall mercury deposition. Depositional rates in south Florida are 30–50% higher than those in central Florida. Measurements of monomethylmercury in precipitation samples ranged from <0.005 to 0.020 ng/L (ppt).

Bowman et al. (2020) analyzed data on mercury levels in the oceans. They determined that greater levels were observed in surface waters (<150 meters) in the Arctic Ocean, but higher levels were observed in deeper layers (150–1,000 m) for other oceans. Average concentrations in the upper surface in the Arctic Ocean ranged from about 1.1 to 1.3 picomoles/L (pM) and from 0.71 to 0.81 pM in the deeper layers. Opposite trends were observed in the Atlantic and Pacific Oceans. For example, the levels in the upper layers of the Pacific Ocean averaged about 0.19–0.99 pM in the <150 m layers and 0.59–1.35 pM in the 150–1,000 m layers. Chen and Li (2019) reported average total mercury levels in the central tropical Pacific Ocean of 0.29–0.54 pM at the thermocline layer (the transition layer between the warmer water at the surface and the cooler deep water below) and 0.98–1.70 pM in deeper layers.

The natural occurrence of mercury in the environment means that mercury is likely to occur in surface waters, even when anthropogenic sources of mercury are absent. Median total mercury levels in 23 streams in the northeastern United States ranged from 0.48 to 10.2 ng/L and median values of methylmercury ranged from below the detection limit of 0.04 ng/L to 0.63 ng/L (Janssen et al. 2019). Methylated mercury levels averaged 0.024 (±0.009 SD) ng/L in Canadian Arctic waters and 0.023 ng/L (±0.011 SD) in the Hudson Bay (Kirk et al. 2012). The USGS conducted sampling from 2012 to 2015 at six locations of the Brownlee Reservoir, Boise, and Snake River in Idaho (USGS 2016). Mercury levels in the water ranged from 0.48 to 8.8 ng/L, with the highest concentration observed in the Brownlee Reservoir. The concentration and speciation of mercury were measured in nine tributaries to Lake Ontario in two independent field sampling programs (Denkenberger et al. 2020). Mean total mercury concentrations were 0.9–2.6 ng/L. Mean total dissolved and particulate phase total mercury levels were 0.5–1.5 and 0.3–2.0 ng/L, respectively in the 9 different tributaries. Mean methylmercury levels were 0.06–0.14 ng/L. Correlations were observed between the levels and speciation of mercury in water and the watershed characteristics and surrounding land use types. For example, total suspended solids in the water column were strongly correlated with the percentage of agricultural land in the watershed and methylmercury as a percentage of total mercury was positively correlated to percent open-water coverage in the watershed.

The baseline concentration of mercury in unpolluted marine waters is typically approximately 0.3 ng/L (Gonzalez-Raymat et al. 2017). In contrast, the New York Bight, an inshore coastal area near the industrialized areas of New York Harbor and northern New Jersey, contained dissolved mercury concentrations in the range of 10–90 ng/L (ppt) (Fowler 1990).

Near-surface groundwaters in remote areas of Wisconsin were found to contain approximately 2–4 ng/L (ppt) of mercury, of which only a maximum of 0.3 ng/L (ppt) was determined to be methylmercury, indicating that groundwater was not a source of methylmercury in the lake (Krabbenhoft and Babiarz 1992). Mercury was found at levels >0.5 μg/L (ppb) in 15–30% of wells tested in some groundwater surveys (EPA 1985). Drinking water is generally assumed to contain <0.025 μg/L (ppb) (EPA 1984). A chemical monitoring study of California's public drinking water from groundwater sources was conducted by Storm (1994). This author reported that mercury was analyzed in 6,856 samples, with 225 positive detections and 27 exceedances of the maximum contaminant level (0.002 mg/L [200 ppb]). The mean mercury concentration was 6.5 ppb (median, 0.62 ppb; range, 0.21 to 300 ppb).

5.5.3 Sediment and Soil

Mercury is a natural constituent of soils occurring at a concentration of approximately 80 ng/g (0.080 ppm) (Gonzalez-Raymat et al. 2017). In a review of the mercury content of virgin and cultivated surface soils from a number of countries, it was found that the average concentrations ranged from 20 to 625 ng/g (0.020–0.625 ppm) (Andersson 1979). The highest concentrations were generally found in soils from urban locations and in organic, versus mineral, soils. The mercury content of most soils varies with depth, with the highest mercury concentrations generally found in the surface layers. A study conducted in 26 European Union countries collected 21,951 topsoil samples (0–20-cm depth) and used a Deep Neural Network (DNN) algorithm to map out the distribution of mercury in European soils (Ballabio et al. 2021). An estimated median level of 38.3 ng/g (0.0383 ppm) was calculated if known contaminated sites were excluded. The study found that mercury concentrations in soil increased with latitude from south to north and with altitude and higher levels were observed around coal use in large power plants. The study authors identified 209 "hotspots" in which soil concentrations exceeded 422 ng/g (0.422 ppm), and nearly half of these (42%) were associated with mining operations.

Median total mercury and methylmercury levels in soil of a remote watershed in the Adirondack Mountains, New York (Fishing Brook) were 170–235 and 0.28–0.94 ng/g (0.170–0.235 and 0.00028–0.00094 ppm), respectively (Burns et al. 2014). The same authors also studied total mercury and methylmercury levels of a coastal watershed (McTier Creek) in South Carolina. Median total mercury and methylmercury levels in these soils were 40–106 and 0.20–1.50 ng/g (0.040–0.106 and 0.00020–0.0015 ppm), respectively. Mercury levels in surface sediment (upper 10 cm) sampled in 10 lakes or watersheds from the Upper Columbia River Watershed in 2012 ranged from 0.01 to 0.28 mg/kg (ppm) (Washington State Department of Ecology 2013). Total mercury levels in 36 samples of soil obtained from a heavily contaminated site near a chloralkali production facility in Romania ranged from 0.08 to 114 mg/kg (ppm), with a mean value of 13.1 mg/kg (ppm; Frentiu et al. 2013). The total mercury concentration in a soil from a polluted chlor-alkali production facility was reported as 1,346 mg/kg (Wang et al. 2023d).

Granato et al. (1995) reported that municipal solid waste sludge mercury concentrations from the Metropolitan Water Reclamation District of Greater Chicago ranged from 1.1 to 8.5 mg/kg (ppm), with a mean concentration of 3.31 mg/kg (ppm). Sludge applications to a sludge utilization site in Fulton County, Illinois, significantly increased extractable soil mercury concentrations from 1971 to 1995. In addition, 80–100% of the mercury applied to the soils in sewage sludge since 1971 still resided in the top 15 cm of soil.

Mercury levels in sediment the Penobscot River, Maine were reported to range from approximately 400 to 1,400 ng/g (range 0.400–1.4 ppm) over a 35-km area, with some samples exceeding 3,000 ng/g (Bodaly

2018). It was estimated that this river received approximately 6–12 metric tons of mercury emitted from a nearby chloralkali plant that was operational from 1967 to 2000. Peat cores obtained from two micro-tidal marshes in the Sacramento-San Joaquin Delta of California were used to reconstruct mercury levels over an approximate 6,000-year period (Drexler et al. 2016). Pre-anthropogenic levels of mercury were estimated to range from approximately 6.9 to 71 ng/g (0.0069–0.071 ppm) and the first man-made sources of mercury introduced to this watershed occurred around 1425 AD. Mercury levels peaked at 990 ng/g around the time of the California gold rush (~1850) as mercury is used to separate gold from its ore by forming an amalgam. A USGS monitoring program in the Great Lakes found that in 2021, 30 sites in Lake Superior had mercury surface sediment levels ranging from 1.1 to 161.2 ng/g and in Lake Huron, mercury sediment concentrations ranged from 0.7 to 113.7 ng/g across all samples (USGS 2023b). Sediment samples collected from Lake Ontario had mercury levels of 19–2,001.1 ng/g (USGS 2023b). The median concentration of total mercury in bed sediments from 23 streams in the northeast United States ranged from 1.3 to 47.6 ng/g normalized to percent organic matter, while the median concentration of methylmercury ranged from 0.04 to 1.8 ng/g normalized to percent organic matter (Janssen et al. 2019). Bulk sediments collected from Galveston Bay, Texas from 2017 to 2019 at four sampling locations had mean levels of 0.01–0.08 μg/g (10–80 ng/g) (Lopez et al. 2022).

Facemire et al. (1995) reported industrial contamination of soils and sediment in several states in the southeastern United States. The study authors reported soil concentrations up to 141,000 ppm (mg/kg) associated with contamination in northeastern Louisiana from mercury-charged manometers (i.e., gas regulators) used to measure pressure and delivery from natural gas wells. In Tennessee, a maximum mercury concentration of 1,100 ppm (associated with previous operations of the Oak Ridge nuclear facility) was found in wetland soils adjacent to the East Fork Poplar Creek. A pharmaceutical company's effluents enriched sediments in a localized area of Puerto Rico to 88 ppm (mg/kg) mercury (Facemire et al. 1995). Rule and Iwashchenko (1998) reported that mean soil mercury concentrations of 1.06 ppm were collected within 2 km of a former chlor-alkali plant in Saltsville, Virginia, and that these concentrations were 17 times higher than regional background soil samples (0.063 ppm [mg/kg]). The study authors further reported that soil organic content, topographic factors, wind patterns, and elevation were variables significantly related to mercury concentration as determined by regression analysis. Soil mercury levels decreasing with distance from the former plant were indicative of a point source distribution pattern. A man-made land soil type (Udorthent), which appears to be a byproduct of the chloralkali manufacturing process, was found proximal to the former plant site and contained about 68 times (4.31 ppm [mg/kg]) the regional background concentration.

A summary of the data for ambient soil/sediment monitoring from the WQP reported in Table 5-16.

Table 5-16. Summary of Concentrations of Total Mercury (ng/g) Measured in Soil and Sediment Across the United States

Year	Average	Maximum	Number of samples	Percent detected
Soil				
2018	33	180	93	94%
2019	120	700	122	21%
2020	96	570	15	100%
2021	100	100	10	40%
2022	100	100	1	100%
2023	0.04	0.05	7	100%
Sediment				
2018	934	79,600	1,676	66%
2019	672	194,000	1,271	73%
2020	588	72,900	2,281	65%
2021	483	18,900	1,105	58%
2022	107	1,600	517	76%
2023	70	184	31	100

Source: WQP 2024

5.5.4 Other Media

Foods. The U.S. Food and Drug Administration (FDA) conducted a Total Diet Study (April 1982 to April 1984) to determine dietary intakes of selected industrial chemicals (including mercury) from retail purchases of foods representative of the total diet of the U.S. population (Gunderson 1988). The data were collected as part of eight food collections, termed Market Basket Surveys collected in regional metropolitan areas during the 2-year study and involved individual analysis of 234 food items representing the diets of eight different population groups. Mercury was detected in 129 adult foods; seafood, the major contributing food group, accounted for 77% (3.01 of the 3.9 µg of mercury) of the total mercury intake for 25–30-year-old males (Gunderson 1988). Minyard and Roberts (1991) reported results of a survey conducted on food samples analyzed at 10 state food laboratories between 1988 and 1989. These laboratories conducted food regulatory programs and analyzed findings of pesticides and related chemical residues for 27,065 food samples. In 1988, these laboratories reported methylmercury residues in 13 (0.09%) of 13,980 samples, with 1 sample exceeding federal or state tolerances. Similarly, in 1989, methylmercury was detected in 25 (0.19%) of 13,085 samples, with 1 sample exceeding federal or state tolerances. A survey of 220 cans of tuna, conducted in 1991 by the FDA, found an average methylmercury content (expressed as mercury) of 0.17 mg/kg (range <0.10–0.75 mg/kg) (Yess 1993). Levels of methylmercury were higher in solid white tuna (0.26 mg/kg) and chunk white tuna (0.31 mg/kg) than in chunk light tuna (0.10 mg/kg) or chunk tuna (0.10 mg/kg). Previously, the FDA had determined methylmercury concentrations in 42 samples of canned tuna between 1978 and 1990 (Yess 1993) to range from <0.01 to 0.67 mg/kg methylmercury (expressed as mercury), with an average concentration of 0.14 mg/kg. These earlier results are similar to those obtained in the 1991 survey (Yess 1993).

Data from the Market Basket Surveys are included in the FDA Total Diet Study for mercury in consumed food items. Data from the 2006–2013 Total Diet Study are shown in Table 5-17. Data from the 2018–2020 Total Diet Study showed that mercury was not detected in most of the samples (245 detects out of 3,276 analyzed samples) (FDA 2022). Only 33 of the 307 different food items tested for mercury had detectable levels. The highest mean mercury concentrations were in fish samples. Canned tuna, baked cod, and baked salmon had mean concentrations of 230 ppb (0.230 mg/kg), 83 ppb (0.083 mg/kg), and 21 ppb (0.021 mg/kg), respectively. Of the 33 foods with detectable results, 28 had mean concentrations <10 ppb (0.010 mg/kg), and all detectable mercury (total mercury) results (mostly seafood) were below the 1 mg/kg (1,000 ppb) action level for methylmercury established for fish, shellfish, crustaceans, and other aquatic animals (FDA 2022). Mercury was only detected in 6% of vegetables analyzed, with a maximum concentration of 1.8 ppb (0.0018 mg/kg) and in 1% of fruit items tested at a maximum level of 1.3 ppb (0.0013 mg/kg). Mercury was not detected in any dairy products from the 2018–2020 survey. It was only detected in 3% of 384 baby food samples and all detections were <3 ppb (0.003 mg/kg).

The use of fish meal as a food for poultry and other animals used for human consumption may result in increased mercury levels in these animals. In Germany, poultry and eggs were found to contain average mercury concentrations of 0.04 and 0.03 mg/kg, respectively. Cattle are able to demethylate mercury in the rumen and thus absorb less mercury; therefore, beef (meat) and cow’s milk contained only 0.001–0.02 and 0.01 mg/kg of mercury, respectively (Hapke 1991). A survey of raw foods in Germany in 1986 found that grains, potatoes, vegetables, and fruits contained average mercury concentrations of 0.005–0.05 mg/kg (fresh weight); however, wild mushrooms contained up to 8.8 mg/kg of mercury. Cocoa beans, tea leaves, and coffee beans contained average mercury concentrations of 0.005, 0.025, and 0.04 mg/kg, respectively. In all cases where the mercury content was high, selenium was also found in measurable, but lower, concentrations (Weigert 1991).

Table 5-17. Mercury Concentrations in Food from the FDA Total Diet Study 2006–2013

Food	Number of analyses	Number of non-detects	Number of trace	Mean (mg/kg)	SD (mg/kg)	Median (mg/kg)	Minimum (mg/kg)	Maximum (mg/kg)	LOD (mg/kg)	LOQ (mg/kg)
Milk, whole, fluid	16	16	0	0	0	0	0	0	0.010	0.040
Milk, lowfat (2%), fluid	16	16	0	0	0	0	0	0	0.010	0.040
Turkey breast, oven-roasted	15	13	2	0.0001	0.0004	0	0	0.001	0.010	0.040
Liver (beef/calf), pan-cooked with oil	15	10	5	0.001	0.001	0	0	0.003	0.010	0.040
Fish sticks or patty, frozen, oven-cooked	15	6	8	0.004	0.004	0.005	0	0.012	0.010	0.040
Eggs, scrambled with oil	14	14	0	0	0	0	0	0	0.010	0.040
Eggs, boiled	16	15	1	0.0001	0.0003	0	0	0.001	0.010	0.040
Peanut butter, smooth/creamy	16	15	1	0.0002	0.0008	0	0	0.003	0.020	0.070
Rice, white, enriched, cooked	16	10	6	0.0004	0.0006	0	0	0.002	0.010	0.040
Oatmeal, plain, cooked	16	15	1	0.0001	0.0005	0	0	0.002	0.010	0.040
Bread, white, enriched	16	15	1	0.0001	0.0005	0	0	0.002	0.020	0.070
Fruit-flavored cereal, presweetened	16	12	4	0.0004	0.0008	0	0	0.002	0.020	0.070
Crisped rice cereal	16	8	8	0.001	0.002	0.001	0	0.004	0.020	0.070
Raisins	16	10	6	0.001	0.001	0	0	0.003	0.020	0.050
Avocado, raw	16	15	1	0.0001	0.0003	0	0	0.001	0.020	0.070
Orange juice, frozen concentrate, reconstituted	14	14	0	0	0	0	0	0	0.010	0.040
Apple juice, bottled	15	15	0	0	0	0	0	0	0.010	0.040
Prune juice, bottled	16	14	2	0.0001	0.0003	0	0	0.001	0.010	0.040
Spinach, fresh/frozen, boiled	16	11	5	0.0003	0.0005	0	0	0.001	0.010	0.040
Collards, fresh/frozen, boiled	16	9	7	0.001	0.001	0	0	0.002	0.010	0.040
Cauliflower, fresh/frozen, boiled	15	15	0	0	0	0	0	0	0.010	0.040
Tomato, raw	16	16	0	0	0	0	0	0	0.010	0.040
Ice cream, light, vanilla	16	16	0	0	0	0	0	0	0.010	0.040
Fruit drink, from powder	16	16	0	0	0	0	0	0	0.010	0.040

Table 5-17. Mercury Concentrations in Food from the FDA Total Diet Study 2006–2013

Food	Number of analyses	Number of non-detects	Number of trace	Mean (mg/kg)	SD (mg/kg)	Median (mg/kg)	Minimum (mg/kg)	Maximum (mg/kg)	LOD (mg/kg)	LOQ (mg/kg)
Baby food, infant formula, milk-based, iron fortified ready to feed (formerly high iron)	16	16	0	0	0	0	0	0	0.010	0.040
Baby food, infant formula, milk-based, low iron, ready to feed	8	8	0	0	0	0	0	0	0.010	0.040
Baby food, chicken and broth/gravy	16	16	0	0	0	0	0	0	0.010	0.040
Baby food, vegetables and beef	16	16	0	0	0	0	0	0	0.010	0.040
Baby food, mixed vegetables	16	16	0	0	0	0	0	0	0.010	0.040
Baby food, pears	16	16	0	0	0	0	0	0	0.010	0.040
Baby food, juice, apple	16	16	0	0	0	0	0	0	0.010	0.040
Yogurt, lowfat, fruit-flavored	16	16	0	0	0	0	0	0	0.010	0.040
Chicken breast, oven-roasted (skin removed)	16	16	0	0	0	0	0	0	0.010	0.040
Chicken nuggets, fast-food	16	16	0	0	0	0	0	0	0.010	0.040
Shrimp, boiled	15	7	8	0.006	0.006	0.007	0	0.016	0.010	0.040
Mushrooms, raw	16	9	7	0.001	0.001	0	0	0.003	0.010	0.040
Tuna noodle casserole, homemade	16	3	9	0.016	0.017	0.011	0	0.067	0.010	0.040
Fish sandwich on bun, fast-food	15	8	7	0.002	0.003	0	0	0.009	0.010	0.040
Clam chowder, New England, canned, condensed, prepared with whole milk	16	12	4	0.0003	0.0004	0	0	0.001	0.010	0.040
Syrup, chocolate	15	13	2	0.0003	0.0008	0	0	0.003	0.010	0.040
Jelly, any flavor	16	15	1	0.0001	0.0003	0	0	0.001	0.010	0.040
Carbonated beverage, fruit-flavored, regular	15	15	0	0	0	0	0	0	0.010	0.040
Baby food, infant formula, soy-based, ready to feed	16	16	0	0	0	0	0	0	0.010	0.040
Baby food, bananas	16	16	0	0	0	0	0	0	0.010	0.040
Salmon, steaks/fillets, baked	16	0	10	0.021	0.009	0.021	0.006	0.039	0.010	0.040

Table 5-17. Mercury Concentrations in Food from the FDA Total Diet Study 2006–2013

Food	Number of analyses	Number of non-detects	Number of trace	Mean (mg/kg)	SD (mg/kg)	Median (mg/kg)	Minimum (mg/kg)	Maximum (mg/kg)	LOD (mg/kg)	LOQ (mg/kg)
Baby food, cereal, rice, dry, prepared with water	15	12	3	0.0003	0.0006	0	0	0.002	0.010	0.040
Baby food, cereal, rice with apples, dry, prepared with water	6	6	0	0	0	0	0	0	0.010	0.040
Chicken breast, fried, fast-food (with skin)	15	15	0	0	0	0	0	0	0.010	0.040
Chicken thigh, oven-roasted (skin removed)	15	15	0	0	0	0	0	0	0.010	0.040
Chicken leg, fried, fast-food (with skin)	15	15	0	0	0	0	0	0	0.010	0.040
Catfish, pan-cooked with oil	15	7	8	0.003	0.004	0.002	0	0.016	0.010	0.040
Tuna, canned in water, drained	15	0	1	0.136	0.114	0.118	0.035	0.509	0.010	0.040
Cranberry juice cocktail, canned/bottled	15	15	0	0	0	0	0	0	0.010	0.040
Beef with vegetables in sauce, from Chinese carry-out	15	12	3	0.0003	0.0006	0	0	0.002	0.010	0.040
Fried rice, meatless, from Chinese carry-out	15	8	7	0.001	0.001	0	0	0.002	0.010	0.040

FDA = U.S. Food and Drug Administration; LOD = limit of detection; LOQ = limit of quantification; SD = standard deviation

Source: FDA 2017a

Pedersen et al. (1994) conducted a monitoring study to assess the levels of trace metals, including mercury, in table wine, fortified wine, beer, soft drinks, and various juices. The study authors reported that in all samples tested, mercury concentrations were at or below the detection limit (6 µg/L).

Fish and Shellfish. Consumption of fish is a major contributor to methylmercury exposure in most populations (EFSA 2012; Kim et al. 2016b; Vejrup et al. 2016; You et al. 2014; Zhang et al. 2010). Mercury levels in Great Lakes aquatic organisms have generally been trending lower since the 1970s; however, since about 1990, they have leveled off or even increased slightly (Grieb et al. 2020; IJC 2015). The initial decrease was likely due to decreasing atmospheric deposition from North America; however, increasing emissions from other parts of the world and climate change, along with many other factors are likely responsible for recent changes (Grieb et al. 2020). The Great Lakes Commission (GLC) reported that fish mercury levels in the Great Lakes region are positively correlated with areas of high forest cover and wetlands because forested areas in these regions of the Great Lakes receive higher dry deposition of mercury and have other watershed features that worsen the impacts of mercury emissions and deposition (BDI 2011). The report also found that fish in waterbodies near agricultural areas tended to have lower mercury levels since increased algal biomass in the aquatic food web tends to reduce methylmercury levels. They also noted that mercury levels in predatory fish such as walleye and largemouth bass are 55 and 25% lower, respectively, in the Great Lakes as compared to inland lakes nearby, which likely occurs due to dissimilarities in the food web structure, land-water linkages, and methylating potential variations between the larger and smaller water bodies. Total mercury levels obtained from fish in 23 streams located in the northeastern United States ranged from 19.5 ng/g (bluegill) to 774.7 ng/g (largemouth bass) (Janssen et al. 2019).

The USGS compiled data from state and federal programs to study the temporal mercury levels in fish in rivers and lakes in the United States from 1969 to 2005 (Chalmers et al. 2011). They observed that declining mercury levels in sediment cores during the 1970s and 1980s correlated with the period of downward mercury levels in fish. Overall, from 1969 to 2005 in 90 rivers and lakes, mercury concentrations had no temporal trends at 57% of the sites, decreasing mercury levels in fish at 32% of the sites, and increasing levels at 11% of the sites. Data from the late 1980s to 2005 showed increasing levels of mercury in fish in some southeastern states, while no trend or decreasing levels in upper midwestern states were observed. These data are summarized in Table 5-18. Another report from the USGS concluded that methylmercury levels in fish exceeded the EPA criterion for protection of human health (0.3 ppm) in predator fish from about 25% of streams sampled nationwide during 1998–2005. Fish methylmercury concentrations tended to be greatest in wetland-dominated streams in the southeastern

United States or in streams draining basins that had been mined for mercury or gold in the West (USGS 2014).

Table 5-18. Trends in Mercury Concentrations Based on Fish Data Aggregated by State from 1988 to 2005[a]

State	Sites	Number	Species	Begin year	End year	Median mercury (mg/kg)	p-Value	Percent change	Trend
Southeastern United States									
Georgia	112	266	LMB	1991	2001	0.24	<0.001	7.43	Up
Georgia	56	105	CCF	1991	2001	0.10	0.063	6.76	None
Louisiana	324	1,049	LMB	1994	2005	0.39	<0.001	3.78	Up
Louisiana	113	168	CCF	1994	2004	0.10	0.007	11.2	Up
Louisiana	178	328	FD	1994	2004	0.37	0.013	6.23	Up
Louisiana	72	125	RS	1994	2004	0.16	0.075	11.4	None
Louisiana	158	383	WC	1994	2004	0.21	0.601	1.14	None
Louisiana	178	378	BC	1995	2004	0.24	0.686	0.83	None
Louisiana	**169**	**444**	**B**	**1994**	**2004**	**0.52**	**0.019**	**-3.26**	**Down**
Louisiana	*42*	*66*	*BMBU*	*1995*	*2004*	*0.28*	*0.045*	*-6.66*	*Down*
North Carolina	37	61	BG	1989	1999	0.10	0.771	0.76	None
South Carolina	**129**	**963**	**B**	**1993**	**2004**	**0.80**	**0.003**	**3.02**	**Up**
South Carolina	70	194	CCF	1994	2004	<0.25	0.277	-2.29	None
South Carolina	**188**	**1,556**	**LMB**	**1993**	**2004**	**0.38**	**<0.001**	**-3.13**	**Down**
Midwestern United States									
Iowa	31	34	LMB	1994	2005	0.13	0.947	0.25	None
Iowa	44	60	CC	1993	2005	0.10	0.005	-6.11	Down
Iowa	87	142	CCF	1988	2005	0.09	<0.001	-6.14	Down
Indiana	194	285	CC	1988	2004	0.17	0.856	-0.11	None
Indiana	56	74	CCF	1988	2004	0.14	0.316	-1.04	None
Indiana	75	91	LMB	1991	2004	0.19	0.999	0.00	None
Michigan	55	158	LMB	1988	1997	0.31	0.632	-0.75	None
Minnesota	43	81	CCF	1990	2000	0.21	0.364	1.80	None
Minnesota	78	157	CC	1990	2001	0.13	0.085	-2.16	None

Table 5-18. Trends in Mercury Concentrations Based on Fish Data Aggregated by State from 1988 to 2005[a]

State	Sites	Number	Species	Begin year	End year	Median mercury (mg/kg)	p-Value	Percent change	Trend
Minnesota	*53*	*142*	*NP*	*1988*	*2001*	*0.27*	*0.001*	*-4.86*	*Down*
Minnesota	75	202	W	1989	2001	0.23	0.036	-3.16	Down

[a] Trends above the U.S. Environmental Protection Agency (EPA) advisory guideline (0.3 μg/g methylmercury) are in bold text. Upward trends starting below and ending above EPA guideline are underlined. Downward trends starting above and ending below EPA guideline are in italics. Trends below EPA guideline are in regular text.

B = bowfin; BC = black crappie; BG = bluegill; BMBU = bigmouth buffalo; CC = common carp; CCF = channel catfish; FD = freshwater drum; LMB = largemouth bass; NP = northern pike; RS = redear sunfish; W = walleye; WC = white crappie

Source: Chalmers et al. 2011

From 2012 to 2015, the USGS collected and analyzed individual fillets of mountain whitefish (*Prosopium williamsoni*), rainbow trout (*Oncorhynchus mykiss*), smallmouth bass (*Micropterus dolomieu*), and channel catfish (*Ictalurus punctatus*) for mercury in the Brownlee Reservoir and Snake River in Idaho (USGS 2016). Mercury levels in rainbow trout were analyzed for one site (Eckert) and had a median concentration of 0.02 mg/kg. Median mercury levels in mountain whitefish were 0.18 mg/kg at the Eckert sampling location and ranged from 0.11 to 0.18 mg/kg at the Middleton site during the 3-year sampling period. Channel catfish collected at three locations had median mercury levels that ranged from 0.11 to 0.28 mg/kg for all three sites. Smallmouth Bass collected from the Brownlee Reservoir in 2013 had median levels of 0.32 mg/kg, which exceeded Idaho water-quality criterion.

Sampling was conducted for different species of fish from 2006 to 2012 in the lower Penobscot River and upper estuary in Maine (Kopec et al. 2019). This river was contaminated by mercury discharges from a chloralkali plant that was operational from 1967 to 2000. Mercury levels were shown to be greatest in fish and shellfish near the plant and downstream from the plant as opposed to upstream or more distant locations. In the most heavily contaminated locations near the chloralkali facility, mean total mercury concentrations in fish muscle adjusted for size or age were 0.521 (95% CI 0.480, 0.566) mg/kg wet weight in American eels, 0.321 (95% CI 0.261, 0.395) in mummichog, 0.121 (95% CI 0.104, 0.140) in rainbow smelt, 0.155 (95% CI 0.142, 0.169) in tomcod, 0.0552 (95% C: 0.0427, 0.0714) in winter flounder, and 0.328 (95% CI 0.259, 0.413) in American lobster tail, and 0.522 (95% CI 0.488, 0.557) mg/kg dw in blue mussel.

Rumbold et al. (2018) studied mercury levels in 50 species of fish at two locations off the coast of the Florida Keys. The first location, Tennessee Reef Lighthouse (TRL), is a bank reef with moderate cover of hard coral, soft-coral, sponge, and macroalgae on a sandy bottom and the second location was a slightly shallower water, Long Key Hard Bottom (LKH), characterized by exposed hard substrate with soft-coral, sponge, and macroalgal cover. These data are presented in Table 5-19.

Table 5-19. Mercury Levels in 50 Species of Fish Obtained at Two Sites Along the Florida Reef Tract from April 2012 to December 2013

Common name	Scientific name	Mean (ng/g)	SD (ng/g)	Maximum (ng/g)	CV (%)
Tennessee Reef Lighthouse					
Bermuda chub	*Kyphosus sectatrix*	78.64	44.87	135.6	57.1
Bicolor damsel	*Stegastes partitus*	26.60	2.96	29.55	11.1
Black grouper	*Mycteroperca bonaci*	347.2	85.95	445.2	24.8
Blue runner	*Caranx crysos*	40.95	NR	NR	NR
Blue stripe grunt	*Haemulon sciurus*	390.9	108.8	506.1	27.8
Blue tang	*Acanthurus coeruleus*	35.28	5.32	44.08	15.1
Brown chromis damsel	*Chromis multilineata*	113.0	NR	NR	NR
Doctorfish tang	*Acanthurus chirurgus*	52.97	7.58	60.98	14.3
French angelfish	*Pomacanthus paru*	43.42	22.11	65.53	50.9
Gray angelfish	*Pomacanthus arcuatus*	58.11	26.34	91.63	45.3
Graysby grouper	*Cephalopholis cruentata*	322.0	68.33	417.4	21.2
Great barracuda	*Sphyraena barracuda*	1,713.8	882.3	3,401.4	51.5
Hogfish	*Lachnolaimus maximus*	129.6	41.44	200.0	31.9
Knobbed porgy	*Calamus nodosus*	153.0	50.23	203.2	32.8
Porkfish	*Anisotremus virginicus*	1,555.1	1,970.9	6,842.3	126.7
Princess parrotfish	*Scarus taeniopterus*	66.84	NR	NR	NR
Red lionfish	*Pterois volitans*	174.9	44.75	225.8	1.7
Redband parrotfish	*Sparisoma aurofrenatum*	42.52	10.17	56.91	23.9
Rock Beauty angelfish	*Holacanthus tricolor*	20.53	1.52	22.04	7.4
Saucereye porgy	*Calamus*	114.6	NR	NR	NR
Schoolmaster snapper	*Lutjanus apodus*	143.1	NR	NR	NR
Scrawled filefish	*Aluterus scriptus*	76.79	6.24	82.57	8.1
Sergeant major damsel	*Abudefduf saxatilis*	57.19	NR	NR	NR
Spanish grunt	*Haemulon macrostomum*	505.9	25.55	531.5	5.0

Table 5-19. Mercury Levels in 50 Species of Fish Obtained at Two Sites Along the Florida Reef Tract from April 2012 to December 2013

Common name	Scientific name	Mean (ng/g)	SD (ng/g)	Maximum (ng/g)	CV (%)
Spanish hogfish	*Bodianus rufus*	300.8	95.90	392.1	31.9
Spotted goatfish	*Pseudupeneus maculatus*	102.2	48.03	184.1	46.9
Stoplight parrotfish	*Sparisoma viride*	30.22	3.33	32.58	11.0
Trumpetfish	*Aulostomus maculatus*	133.3	NR	NR	NR
White grunt	*Haemulon plumierii*	304.1	53.05	340.1	10.6
Yellowtail damsel	*Chrysiptera parasema*	35.91			
Yellowtail snapper	*Ocyurus chrysurus*	104.4	45.45	149.8	43.5
Long Key Hard Bottom					
Atlantic spadefish	*Chaetodipterus faber*	242.6	66.47	310.9	27.4
Bermuda chub	*Kyphosus sectatrix*	50.99	23.80	97.41	46.7
Black grouper	*Mycteroperca bonaci*	343.1	13.14	356.3	3.8
Blue runner	*Caranx crysos*	252.2	166.0	460.7	65.8
Blue stripe grunt	*Haemulon sciurus*	265.4	65.28	356.1	24.6
Blue tang	*Acanthurus coeruleus*	40.90	NR	NR	NR
Cocoa damsel	*Stegastes variabilis*	118.4	NR	NR	NR
Doctorfish tang	*Acanthurus chirurgus*	96.37	NR	NR	NR
French angelfish	*Pomacanthus paru*	53.31	NR	NR	NR
Gray angelfish	*Pomacanthus arcuatus*	17.49	0.16	17.65	0.9
Gray snapper	*Lutjanus griseus*	190.2	31.10	216.1	16.4
Graysby grouper	*Cephalopholis cruentata*	152.5	NR	NR	NR
Great barracuda	*Sphyraena barracuda*	3,317.5	NR	NR	NR
Hogfish	*Lachnolaimus maximus*	142.3	55.66	257.5	39.1
Horse-eye jack	*Caranx latus*	147.8	NR	NR	NR
Lane snapper	*Lutjanus synagris*	256.2	NR	NR	NR
Leatherjacket	*Oligoplites saurus*	747.7	43.99	791.7	5.9
Lookdown	*Selene vomer*	234.9	90.64	322.3	38.5
Mutton snapper	*Lutjanus analis*	196.9	NR	NR	NR
Ocean surgeon	*Acanthurus bahianus*	41.41	0.02	41.43	0.0
Pigfish	*Orthopristis chrysoptera*	857.5	NR	NR	NR
Planehead filefish	*Stephanolepis hispidus*	77.57	NR	NR	NR
Porkfish	*Anisotremus virginicus*	501.6	151.1	702.5	30.1
Red grouper	*Epinephelus morio*	197.8	24.77	232.8	12.5
Red lionfish	*Pterois volitans*	239.3	25.01	264.4	10.4
Rock hind grouper	*Epinephelus adscensionis*	144.0	9.35	153.4	6.5

Table 5-19. Mercury Levels in 50 Species of Fish Obtained at Two Sites Along the Florida Reef Tract from April 2012 to December 2013

Common name	Scientific name	Mean (ng/g)	SD (ng/g)	Maximum (ng/g)	CV (%)
White grunt	*Haemulon plumierii*	320.5	78.05	494.7	24.3
Yellow jack	*Carangoides bartholomaei*	236.4	129.6	366.0	54.8
Yellowtail snapper	*Ocyurus chrysurus*	142.5	38.52	223.2	27.0

CV = coefficient of variation (SD/mean); NR = not reported; SD = standard deviation for means based on ≥2 fish

Source: Rumbold et al. 2018

From 1986 to 1989, the National Study of Chemical Residues in Fish (NSCRF) was conducted by the EPA to assess the concentrations of 60 toxic pollutants (including mercury) in the tissues of benthic and predatory gamefish nationwide (EPA 1992). Benthic species were analyzed as whole-body samples, while game fish species were analyzed as fillet samples, and all concentrations were reported on a wet weight basis. Mercury was detected at 92% of the 374 sites surveyed nationwide at a mean concentration of 0.260 mg/kg (median concentration of 0.17 mg/kg and maximum concentration of 1.8 mg/kg), and at 2% of the sites, measured mercury concentrations exceeded 1 mg/kg. Most of the higher mercury concentrations in fish were collected in the Northeast. Ten of the sites in the top 10th percentile for high mercury concentrations were near pulp and paper mills, four were near Superfund sites, and most of the remaining sites were near industrial areas. However, the mercury sources could not be identified at all of these sites. Five sites were considered to represent background conditions and six USGS National Stream Quality Accounting Network (NASQAN) sites were also among the sites in the top 10th percentile (EPA 1992).

A national survey conducted by the EPA solicited data on mercury concentrations in fish collected by the states as part of their fish contaminant monitoring programs (EPA 1999b). The EPA asked all states to submit mercury residue data collected from their fish sampling programs from 1990 through 1995 to assess whether there were geographic variations or trends in fish tissue concentrations of mercury. Thirty-nine states provided information on the levels of contamination in their fish. The study included the following: information on the tissue concentrations of mercury, including the number of fish sampled (by species); the mean mercury concentration; and the minimum, median, and maximum concentrations reported for each species by state. Residue information for the three most abundant species sampled in each state included such species as the largemouth and smallmouth bass; channel, flathead, and blue catfish; brown and yellow bullhead; rainbow and lake trout; carp; walleye; north pike; and white sucker.

A summary of the mean, minimum, and maximum tissue concentrations of mercury detected for two of the sampled species with the widest geographical distribution; the largemouth bass and the channel catfish are given in Tables 5-20 and 5-21, respectively. As Table 5-20 shows, the maximum mercury residues reported for the largemouth bass exceeded the FDA action level (1 ppm [1 mg/kg]) in 16 states that collected and analyzed tissue samples for this species. The highest maximum mercury concentration reported for this species was 8.94 mg/kg, reported by New Jersey. Table 5-21 shows the maximum mercury residue reported for another widely distributed species, the channel catfish. While the maximum mercury residues reported for this species are not consistently as high as those for the largemouth bass, maximum residues in channel catfish from six states still exceeded the FDA action level (1 ppm [1 mg/kg]). Consumption of large amounts of feral fish containing these high mercury residues exposes high volume fish consuming populations (those that consume >100 g fish/day) to potentially greater risk of mercury exposure than members of the general population.

Table 5-20. Mercury Concentrations (mg/kg) for Largemouth Bass Collected in Various States Throughout the United States (1990–1995)

State	Number of fish	Minimum	Mean[a]	Maximum[b]
Alabama	914	0.100	0.393	**1.630**
Arizona	35	0.700	**1.369**	**2.620**
Arkansas	1,190	0.030	0.675	**3.170**
California	517	0.030	0.291	**1.800**
Connecticut	507	0.032	0.505	**2.645**
District of Columbia	11	0.037	0.153	0.458
Florida	2,000	0.020	0.645	**4.360**
Georgia	968	0.010	0.274	**2.286**
Illinois	305	0.010	0.018	0.880
Louisiana	452	0.001	0.391	**1.883**
Maine	137	0.071	0.634	**1.343**
Massachusetts	152	0.045	0.399	**1.100**
Mississippi	505	0.090	0.651	**2.630**
Missouri	106	0.002	0.257	0.608
Nebraska	182	0.080	0.343	0.920
New Hampshire	35	0.210	0.573	**1.400**
New Jersey	173	0.030	0.664	**8.940**
New York	53	0.050	0.462	0.950
North Carolina	1,569	0.020	0.532	**3.600**

Table 5-20. Mercury Concentrations (mg/kg) for Largemouth Bass Collected in Various States Throughout the United States (1990–1995)

State	Number of fish	Minimum	Mean[a]	Maximum[b]
Oregon	140	0.030	0.332	0.980
Pennsylvania	139	0.090	0.293	0.750
South Carolina	505	0.230	0.994	**3.330**
Tennessee	64	0.100	0.255	0.830
Texas	58	0.043	0.237	0.657
Vermont	1	0.150	0.802	**1.200**
Washington	20	0.024	0.137	0.350

[a] Weighted average of composite samples where the weight is the number of fish in each composite ($\Sigma(C_i \times N_i)/N_t$, where C_i and N_i are the concentrations and number of fish in each composite sample, respectively, and N_t is the total number of fish in all composites).
[b] Tissue concentrations shown in **bold type** exceed the U.S. Food and Drug Administration (FDA) action level of 1 ppm (1 mg/kg).

Source: EPA 1999b

Table 5-21. Mercury Concentrations (mg/kg) for Channel Catfish Collected in Various States Throughout the United States (1990–1995)

State	Number of fish	Minimum	Mean[a]	Maximum[b]
Alabama	702	0.100	0.214	0.660
Delaware	19	0.020	0.050	0.133
District of Columbia	17	0.055	0.091	0.240
Georgia	658	0.010	0.084	**1.1143**
Iowa	323	0.030	0.104	0.410
Kansas	56	0.029	0.125	0.314
Louisiana	76	0.001	0.111	0.732
Maryland	157	0.006	0.033	0.256
Michigan	964	0.014	0.047	0.710
Mississippi	157	0.040	0.272	2.100
Missouri	198	0.002	0.052	0.350
Nebraska	238	0.001	0.109	0.643
New Mexico	78	0.100	0.297	**1.800**
Ohio	574	0.018	0.118	1.040
Oklahoma	324	0.100	0.193	0.640
South Carolina	42	0.250	0.345	**1.610**
Tennessee	138	0.100	0.173	0.650

Table 5-21. Mercury Concentrations (mg/kg) for Channel Catfish Collected in Various States Throughout the United States (1990–1995)

State	Number of fish	Minimum	Mean[a]	Maximum[b]
Texas	44	0.043	0.193	**1.186**
West Virginia	65	0.030	0.139	**1.583**

[a] Weighted average of composite samples where the weight is the number of fish in each composite ($\Sigma(C_i \times N_i)/N_t$, where C_i and N_i are the concentrations and number of fish in each composite sample, respectively, and N_t is the total number of fish in all composites).
[b] Tissue concentrations shown in **bold type** exceed the U.S. Food and Drug Administration (FDA) action level of 1 ppm (1 mg/kg).

Source: EPA 1999b

The Northeast states and Eastern Canadian provinces issued their own mercury study, including a comprehensive analysis of current mercury concentrations in a variety of freshwater sportfish species (NESCAUM 1998). This study involved a large number of fish sampling sites in each state, many of which were remote lake sites that did not receive point source discharges. Top level piscivores (i.e., predatory fish) such as walleye, chain pickerel, and large and smallmouth bass were typically found to exhibit some of the highest concentrations, with average tissue residues >0.5 mg/kg and maximum residues >2 mg/kg. One largemouth bass sample was found to contain 8.94 mg/kg of mercury, while one smallmouth bass sampled contained 5.0 mg/kg. A summary of the mean and minimum–maximum (range) of mercury concentrations in eight species of fish sampled is shown in Table 5-22. This study also identified a relationship between elevated mercury levels in fish and certain water quality parameters, including low pH, high conductivity, and elevated levels of dissolved organic carbon.

Table 5-22. Combined Data on Mercury Concentrations in Selected Fish Species Sampled in the Northeast[a]

Species	Number of samples[b]	Mean mercury concentration	Minimum–maximum mercury[c] concentration range (mg/kg)
Largemouth bass	1,019	0.51	0–**8.94**
Smallmouth bass	738	0.53	0.08–**5.0**
Yellow perch	1,346	0.40	0–**3.15**
Eastern chain pickerel	157	0.63	0–**2.81**
Lake trout	877	0.32	0–**2.70**
Walleye[d]	257	0.77	0.10–**2.04**

Table 5-22. Combined Data on Mercury Concentrations in Selected Fish Species Sampled in the Northeast[a]

Species	Number of samples[b]	Mean mercury concentration	Minimum–maximum mercury[c] concentration range (mg/kg)
Brown bullhead	421	0.20	0–**1.10**
Brook trout	200	0.26	0–0.98

[a] Northeastern states include Main, Vermont, New Hampshire, Massachusetts, Rhode Island, Connecticut, New York, and New Jersey.
[b] In some cases, states reported an average of values from a given location; thus, the number of samples indicated may not represent the number of individual fish sampled.
[c] Maximum tissue concentrations shown in **bold type** exceed the U.S. Food and Drug Administration action level of 1 mg/kg (1 ppm).
[d] Walleye data are from New York State only and may not be representative of walleye mercury concentrations in other parts of the northeast.

Source: NESCAUM 1998

Methylmercury constitutes over 99% of the total mercury detected in fish muscle tissue, with no detection of inorganic or dimethylmercury (Bloom 1992; Grieb et al. 1990). Mercury levels were examined in aquatic organisms taken from the Calcasieu River/Lake Complex in Louisiana. The order of enrichment was as follows: shrimp (0.2 mg/kg) < mussel (0.3 mg/kg) < fish (0.4 mg/kg) = oyster (0.4 mg/kg) < zooplankton (1.4 mg/kg) (Ramelow et al. 1989). Average mercury concentrations for aquatic organisms collected from the Wabigoon/English/Winnipeg River system in Canada were as follows: 0.06–2.2 mg/kg for crayfish, 0.01–0.55 mg/kg for perch, and 0.04–1.2 mg/kg for pike. Methylmercury concentrations were found to increase with distance from the pollutant source, possibly as a result of the increased bioavailability of organic mercury produced by aquatic microorganisms, whereas inorganic mercury was the predominant form at the source (Parks et al. 1991).

In a study of sportfish collected in San Francisco Bay, Fairey et al. (1997) reported that the highest concentrations of mercury were detected in leopard shark muscle tissue (1.26 mg/kg). Bluefin tuna caught in the Northwest Atlantic Ocean in 1990 contained mercury at a mean muscle concentration of 3.41 mg/kg dry weight (Hellou et al. 1992). Arctic cod (*Arctogadus glacialis*) that were obtained during the spring season under ice layers in the Amundsen Gulf/Franklin Bay had average mercury levels of 0.37 mg/kg dry weight and were significantly higher than those collected from the shallow coastal shelf region of the Beaufort Sea, near the Mackenzie Delta, suggesting differences in regional food webs (Kirk et al. 2012).

Marine Mammals. Consumption of marine mammals can be an important source of exposure to methylmercury in populations that are high consumers of marine mammals (Grandjean et al. 1992). Mercury concentrations have been analyzed in various tissues (i.e., muscle, liver, kidneys) from several species of marine mammals, including beluga whales, narwhal, white-toothed dolphins, pilot whales, ringed seals, harp seals, and walruses in the western and eastern Canadian Arctic (Wagemann et al. 1995). The mean mercury concentration (mg/kg dry weight) in liver tissue was highest in pilot whales (78 mg/kg), harp seals (36 mg/kg), Eastern Arctic ringed seals (29 mg/kg), narwhal (25 mg/kg), and Eastern Arctic beluga (22 mg/kg), with lesser amounts in Arctic walrus (5 mg/kg) and dolphins (4 mg/kg). Of the three tissues analyzed, mercury was most concentrated in the liver, with successively lower concentrations in the kidney and muscle tissue. This pattern prevails in most marine mammals. The concentration of total mercury is greater by a factor of 3 in the liver than in the kidney but can be significantly higher in some species (Table 5-23). Mean tissue residues in ringed seals from the western Arctic had significantly higher concentrations of mercury than those from the eastern Arctic. The study authors reported higher mercury levels in sediment (0.068–0.243 mg/kg dry weight) and water (11–29 ng/L) from the western Arctic, as compared to sediment (0.040–0.060 mg/kg dry weight) and water (3.7 ng/L) from the eastern Arctic. These differences in sediment and water mercury levels may be responsible for some of the observed differences in mercury tissue concentrations in the seals.

Table 5-23. Total Mercury Concentrations in Tissues of Marine Mammals in Alaska and Canada (mg/kg, Wet Weight)

Species	Date collected (location)	Muscle concentration (n)	Liver concentration (n)	Source
Polar bear	1972 (West Alaska)	0.043±0.001 (16)	4.235±1.385 (25)	ADFG 1976
	1972 (North Alaska)	0.168±0.089 (30)	29.914±22.547 (38)	
Beluga whale	1977 (South Beaufort Sea)	2.12±0.15 (11)	30.62±20.53 (8)	Muir et al. 1992
Ringed seal	1972 (Southeast Beaufort Sea)	0.23±0.11 (13)	1.0±1.16 (13)	Smith and Armstrong 1975
	1972–1973 (Amundsen Gulf)	0.72±0.33 (83)	27.50±30.10 (83)	Smith and Armstrong 1975, 1978
	1976 (Barrow Strait)	0.91±0.38 (27)	16.14±13.84 (27)	Smith and Armstrong 1978
	1976 (Strathcona Sound)	0.08±0.07 (37)	0.32±0.080 (36)	
	1976 (North Baffin Island)	0.31±0.17 (33)	3.76±3.42 (33)	

Table 5-23. Total Mercury Concentrations in Tissues of Marine Mammals in Alaska and Canada (mg/kg, Wet Weight)

Species	Date collected (location)	Muscle concentration (n)	Liver concentration (n)	Source
Bearded seal	1973 (Amundsen Gulf)	0.53±0.35 (3)	143±170 (6)	Smith and Armstrong 1975, 1978
	1974 (East Hudson Bay)	0.09±0.04 (55)	26.18±26.13 (56)	Smith and Armstrong 1978

Mercury tissue concentrations were detected in 17 adult and 8 fetal pilot whales from two stranding episodes off Cape Cod, Massachusetts (Meador et al. 1993). Total mercury occurred in high concentrations in both the liver and kidney, and liver concentrations were significantly correlated with the animal's length. Methylmercury, as a percentage of total mercury, varied inversely with total mercury, indicating that demethylation was occurring. Mean adult mercury concentrations in mg/kg dry weight in liver and kidneys were 176 mg/kg (range 1.9–626 mg/kg dry weight) and 27.5 mg/kg (range 6.8–49.7 mg/kg dry weight), respectively. Mean fetal mercury concentrations in mg/kg dry weight in liver and kidneys were 2.3 mg/kg (range 0.9–5.4 mg/kg dry weight) and 1.9 mg/kg (range, 0.6–3.9 mg/kg dry weight), respectively. The mean methylmercury concentration in mg/kg dry weight in adult liver tissue was 8 mg/kg (range 5.6–10 mg/kg). Aguilar and Borrell (1995) studied mercury tissue levels (1970–1988) in harbor porpoises in the eastern North Atlantic. The study authors reported that in most tissues of harbor porpoises, the mercury was virtually all in the form of methylmercury; however, the fraction of organic mercury in the liver was much lower than in the rest of the body tissues. The study authors found that for a given tissue, the concentrations detected were extremely variable between localities and years. Mercury concentrations in harbor porpoises ranged from 0.62 to 70 mg/kg in liver and from 0.66 to 22 mg/kg in muscle. The mean mercury concentration in liver for the eastern harbor porpoise population was 11.2 mg/kg. Mercury tissue levels progressively increased with the age of the animal; no significant differences were found between the sexes (Aguilar and Borrell 1995).

Plants. Consumption of rice can also make a substantial contribution to dietary mercury intake, and, in some populations, rice has been shown to be the dominant sources of dietary mercury intake (Zhang et al. 2010). Rice is a particularly susceptible crop for mercury accumulation since it grows in wet, often flooded, anaerobic conditions, which are favorable to the transformation of elemental mercury into methylmercury (Sizmur et al. 2018). Mortimer (1985) reported that total mercury in the roots of five species of freshwater vascular plants in the polluted Ottawa River was 10–40% higher than in the shoots. Speciation may be important in determining the patterns of mercury uptake, translocation, and excretion in macrophytes. Shoots of *Elodea densa* more readily accumulated methylmercury than inorganic mercury, and also excreted more inorganic mercury than methylmercury (Czuba and Mortimer 1980). Significant translocation of inorganic mercury from shoots to roots occurred in *E. densa* (Czuba and Mortimer 1980). In this species, methylmercury and inorganic mercury moved in opposite directions, with methylmercury moving towards the young shoot apex and inorganic mercury moving towards lower (older) parts of the shoot (Czuba and Mortimer 1980). Dolar et al. (1971) noted the same methylmercury pattern in the water milfoil (*Myriophyllum spicatum*). Using solution culture experiments, the study authors showed that mercury accumulation was greater when plants were exposed to inorganic mercury ($HgCl_2$) than organic methylmercury (CH_3HgCl) and that mercury accumulation from the nutrient solution was rapid and approached maximum values in 2 hours. Organomercury compounds (methylmercury chloride, phenylmercuric acetate, phenylmercuric chloride, and phenylmercuric hydroxide) were more available than inorganic compounds (HgF_2 and $HgCl_2$) from lake sediments. The various organomercury and inorganic mercury compounds were added to sediment at concentrations of 0, 46, 230, and 460 mg/kg prior to rooting water milfoil. After 20 days, concentration of mercury in the plant tissues exposure to 46, 230, and 460 mg/kg of the inorganic mercury compounds in the sediment were 1.71–4.01, 4.81–6.03, and 6.61–10.2 mg/kg, respectively. In contrast, the concentrations of mercury in plant tissues exposed to 46, 230, and 460 mg/kg of the organic mercury compounds in the sediment were 2.40–7.15, 36–84.5, and 114.6–243.1 mg/kg, respectively. The control plants (no mercury compounds added to the sediments) contained 0.3 mg/kg mercury. It is clear from this experiment that organomercury compounds may accumulate significantly in the above-ground parts of some macrophytes. Mortimer (1985) found that although *E. densa* shoots had lower total mercury contents than roots, with 32% of the mercury in the shoots in the form of methylmercury, compared to only 10% in the roots.

Grasses sampled downwind of a municipal waste incinerator contained up to 0.20 μg/g (ppm) of mercury, with concentrations decreasing with increasing distance from the facility (Bache et al. 1991). Background mercury levels in vegetation were usually <0.1 mg/kg dry weight (Lindqvist et al. 1991); however, mushrooms collected 1 km from a lead smelter in Czechoslovakia contained between 0.3 and 12 mg/kg dry weight (Kalač et al. 1991).

Consumer and Medicinal Products. Various consumer and medicinal products may contain mercury or mercury compounds (e.g., skin lightening creams and soaps, herbal remedies, laxatives, tattooing dyes, fingerpaints, artists paints, and make-up paints), but all of these products originate from outside of the United States (Barr et al. 1973; DeVito and Brooks 2013; Dyall-Smith and Scurry 1990; Lauwerys et al. 1987; McKelvey et al. 2011; Rastogi 1992; Wendroff 1990). The EPA maintains a website that lists the consumer products that may contain mercury (https://www.epa.gov/mercury/mercury-consumer-products#list).

Barr et al. (1973) reported elevated mercury levels in the blood of women using skin lightening creams, although the mercury compound and concentrations in the skin cream were not determined. Dyall-Smith and Scurry (1990) reported that one skin lightening cosmetic cream contained 17.5% mercuric ammonium chloride. Lauwerys et al. (1987) reported a case of mercury poisoning in a 3-month-old infant whose mother frequently used a skin lightening cream and soap containing inorganic mercury during her pregnancy and during the 1-month lactation period following birth. However, the mercury concentration and specific mercury compound in the cream and soap were not determined. Al-Saleh and Al-Doush (1997) analyzed the inorganic mercury content of 38 skin lightening creams in Saudi Arabian markets. The creams were manufactured in a variety of countries, including India and Pakistan, other Arab countries, Thailand, Taiwan, Indonesia, England, and Germany. Almost 50% of the creams tested exceeded the tolerance limit of 1 ppm. The mean concentration of mercury in the 38 creams was 994 mg/kg, with a range of 0–5,650 mg/kg. It is not known whether any of these products are available in the United States.

Metallic mercury was also the source of two cases of mercury poisoning caused by the dermal application of an over-the-counter anti-lice product (Bourgeois et al. 1986). The more severely poisoned individual applied 30 g of ointment containing 9 g of metallic mercury (300,000 mg/kg) to his entire body. Wands et al. (1974) also reported the deaths of two individuals due to the excessive use of a laxative preparation containing mercurous chloride (calomel).

Metallic mercury has been used by Mexican-American and Asian populations in traditional remedies for chronic stomach disorders (Espinoza et al. 1995, 1996; Geffner and Sandler 1980; Trotter 1985). Perharic et al. (1994) reported cases of poisonings resulting from exposure to traditional remedies and food supplements reported to the National Poisons Unit in London, England. From 1989 to 1991, elemental mercury was implicated in several poisonings following exposure to traditional Asian medicines. In one case, the mercury concentration in the medicinal product taken orally was 540,000 mg/kg. The mercury was in its elemental or metallic form. Espinoza et al. (1995, 1996) reported that while examining imported Chinese herbal balls for the presence of products from endangered species, the study authors detected potentially toxic levels of arsenic and mercury in certain herbal ball preparations. Herbal balls are aromatic, malleable, earth-toned, roughly spherical, hand-rolled mixtures primarily composed of herbs and honey that are used to make medicinal teas. These herbal balls are used as a self-medication for a wide variety of conditions, including fever, rheumatism, apoplexy, and cataracts. Herbal balls similar to those analyzed are readily available in specialty markets throughout the United States. Mercury (probably mercury sulfide) was detected in eight of the nine herbal balls tested. The recommended adult dose for the herbal balls is two per day. Ingesting two herbal balls could theoretically provide a dose of up to 1,200 mg of mercury.

Samudralwar and Garg (1996) conducted trace metal analysis on a variety of plants used in Indian herbal remedies and other medicinal preparations. The study authors reported mercury concentrations of 0.139, 0.180, 0.027, 0.0125, 0.0117, and <0.010 mg/kg for Bowen's kale, Neem leaves, Gulvei leaves, Kanher bark, Vekhand root, and orange peel, respectively.

Hoet and Lison (1997) reported on an unusual non-occupational source of mercury exposure in a woman who used prescription nasal drops that contained 300 mg/L borate phenylmercury. The study authors reported that the woman, who had used the nasal drops over a long period of time, had high urinary levels of mercury (82 mg/kg), but that blood levels were not abnormal (5.5 μg/L).

Mercuric sulfide, or cinnabar, was reported to be used in tattooing dyes to produce a red pigmentation (Bagley et al. 1987; Biro and Klein 1967). An analysis of finger paints and make-up paints manufactured in Europe showed that they all contained <1 mg/kg mercury (Rastogi 1992). Rastogi and Pritzl (1996) conducted another study to assess the migration of several toxic metals from crayons, watercolor paints, and water-based paints. Migration of mercury from the art materials was determined by scraping flakes of the products into dichloromethane for 2 hours at 54°C. The degreased material was then placed in an aqueous HCl solution, shaken, and centrifuged. The supernatant was then filtered through a 0.45 μm membrane filter and analyzed. The study authors reported that the migration of mercury from these art supplies was 0.24–5.98 mg/kg for red paint, 0.26–3.63 mg/kg for blue paint, 0.20–4.79 mg/kg for yellow paint, 0.22–5.68 mg/kg for green paint, and 0.17–3.63 mg/kg for white paint. Migration of mercury from the product occurred in 57% of the samples tested. The migration limit set by European Standard EN71-3 for mercury is 60 mg/kg. This value was not exceeded in any of the art supplies tested. The study authors, however, believe that children might be exposed not only to mercury, but to several other metals that also co-migrated from the paints. Mercury is not allowed in the manufacture of any paint products in the United States (DeVito and Brooks 2013).

Cigarettes. In a study conducted in West Germany, Pesch et al. (1992) analyzed mercury concentrations in 50 brands of cigarettes manufactured in two Western and six Eastern European countries. The study authors reported that in 1987, the average mercury concentration detected in cigarettes was 0.098 mg/kg (dry weight) (range 0.06–0.14 mg/kg dry weight). In 1991, the mean mercury concentrations for cigarettes were 0.034 mg/kg dry weight (range 0.007–0.092 mg/kg dry weight) for Eastern Europe and 0.015 mg/kg dry weight (range 0.006–0.037 mg/kg dry weight) for Western European countries. The study authors attributed the decline in mercury content of cigarettes to environmental protection measures instituted in the intervening years (Pesch et al. 1992).

Religious and Cultural Rituals, Ceremonies, and Practices. Some practitioners of religious, folk, cultural, or ritualistic practices such as Santeria, Voodoo, Palo Mayombe, and Espiritismo have used mercury in their practices (WHO 2010). In the United States, people may obtain metallic mercury (sometimes under the name azogue) in shops called botanicas. Botanicas typically dispense mercury in gelatin capsules or sometimes in small glass vials.

Some people carry a small amount of mercury in a vial, or mix mercury in bath water or perfumed soaps, devotional candles, ammonia, or camphor. Other people's religious practices involve sprinkling metallic mercury on the floor of a dwelling or car, mixing metallic mercury with soap and water to wash the floor, or placing it in an open container to rid the house of evil spirits. Any of these practices can liberate mercury vapor into the room air, exposing the occupants to elevated levels of mercury vapors (ATSDR 1997; Wendroff 1990, 1991). This use of mercury can contaminate a dwelling or automobile if the mercury is not completely removed from flooring, carpeting, and woodwork in an appropriate manner.

In addition to the individuals who intentionally use mercury in their dwellings, the opportunity exists for non-users to be inadvertently exposed when they visit the dwelling, or purchase or rent dwellings in which the former tenants used mercury for religious purposes (NJDEP 2007; Riley et al. 2006). In one study, mean mercury levels were significantly ($p<0.05$) elevated at 9.8 ng/m^3 in 60 building common areas with suspected cultural mercury use. The referent community buildings ($n=40$) recorded 5.0 ng Hg/m^3 in their common areas. Likewise, the maximum mercury levels in buildings with suspected cultural mercury use were also significantly elevated compared to the referent community buildings, with values of 13.3 and 6.4 ng Hg/m^3, respectively (Garetano et al. 2008). The issuance of cautionary notices and information by health departments to members of these user populations is appropriate (Rogers et al. 2007).

5.6 GENERAL POPULATION EXPOSURE

Humans can be exposed to mercury in air, water, soil, and food. Diet is typically the major source of mercury absorption in the general population (non-occupational). However, mercury released from mercury amalgam dental restorations can also contribute to mercury absorption (Langworth et al. 1988; Mackert and Berglund 1997; Nylander et al. 1987). The dominant source of mercury intake and absorption from the diet derives from consumption of fish (Davis et al. 2014; De Winter-Sorkina et al. 2003; EFSA 2012; Kim et al. 2016b; Lescord et al. 2018;), in part because of the relative high concentrations of methylmercury in fish and shellfish (Bloom 1992; Lescord et al. 2018; Storelli et al. 2003; Wells et al. 2020) and near complete absorption of methylmercury in the human gastrointestinal tract (Section 3.1 Toxicokinetics).

Consumption of rice can also make a substantial contribution to dietary mercury intake and, in some populations, rice has been shown to be the dominant sources of dietary intake (Rothenberg et al. 2016b; Sizmur et al. 2018; Wells et al. 2020; Zhang et al. 2010). Dietary mercury intake has been estimated from food surveys in various populations and, as can be expected, varies with diet and, in particular, the contribution of fish and shellfish and rice to the total diet (De Winter-Sorkina et al. 2003; EFSA 2012; Kim et al. 2016b; Vejrup et al. 2016; WHO 1990; You et al. 2014; Zhang et al. 2010). Total diet studies conducted in Asia, United States, and Europe suggest that intakes of total mercury ranging from 1 to 10 μg/day are typical (Carrington and Bolger 2002; EFSA 2014; EPA 1999b; Jenssen et al. 2012; Kim et al. 2016b; Sanga et al. 2001; WHO 1990). Intakes can be substantially higher in populations that consume higher amounts of fish (Dong et al. 2015; Juric et al. 2017; Marien and Patrick 2001; Passos et al. 2008).

The relative contribution of mercury released from dental amalgams has been estimated based on studies of release rates and assumptions regarding the fate and absorption of amalgam mercury. The contributions of mercury from dental amalgams were estimated based on results from measurements of releases of Hg^0 vapor and particulate Hg^0 from amalgams and models of intake and absorption of mercury released from amalgams (Mackert and Berglund 1997). Total mercury absorption in a person having 13 mercury amalgam dental restorations was estimated to be approximately 3 μg/day (range 0.6–9.3 μg/day), of which approximately one-third was absorbed from the gastrointestinal tract. Following removal of all dental amalgams, mean blood total mercury decreased 1.13 μg/L (SD 0.6) from a baseline of 2.18 μg/L (SD 0.90) over an 18-week period, a 49% decrease, in subjects who had an average of seven occlusal surfaces with amalgam restorations (Snapp et al. 1989). This would be consistent with amalgams contributing to approximately half of total mercury absorption. Removal of mercury amalgams can release Hg^0 to air and saliva (Halbach et al. 2000; Warwick et al. 2019).

The general population may also be exposed to mercury vapor released from liquid elemental mercury from breakage or spills of older mercury-containing items at home or at school, such as medical/scientific equipment, batteries, fluorescent lamps, electrical switches, and paints made prior to 1992 (CDC 2015). Disposal of older items and proper clean-up of spills reduces the change of exposure (CDC 2013, 2015).

Various consumer (e.g., skin lightening creams and soaps, herbal remedies, laxatives, tattooing dyes, fingerpaints, artists paints, and make-up paints) and medicinal products (e.g., thimerosal, an ethylmercury-containing compound that was used as a preservative in vaccines) that contain mercury or mercury compounds can also contribute to exposure to consumers (DeVito and Brooks 2013; McKelvey et al. 2011; Rastogi 1992; Wendroff 1990). Any mercury released into air, water, or soil via consumer use or disposal of mercury-containing products would contribute to exposures detected in environmental media.

Mercury levels in blood and urine are measured as part of the NHANES (CDC 2024) (Tables 5-24–5-33). Based on survey data for the period 2017–2018 (the most recent data available in CDC 2024), the geometric mean total BHg level in the adult U.S. population was estimated to be 0.730 µg/L (95% CI 0.620, 0.840). The geometric mean methylmercury blood level was 0.500 µg/L (95% CI 0.420, 0.610). Total and methylmercury blood levels in young children were lower than in adults. The 50th percentiles for total BHg levels in children 1–5 years of age were less than the detection limit (0.28 µg/L) in 2017–2018. For the 2011–2012 period, the detection limits for total mercury were lower, reporting a geometric mean total BHg level of 0.262 µg/L (95% CI 0.237, 0.291) in children 1–5 years of age. The 50th percentiles for methylmercury blood levels in children 1–5 years of age were less than the detection limit (0.12 µg/L) during both time periods. For the 2017–2018 period, the 50th percentiles of total urinary mercury were below the detection limit (0.13 µg/L) in children 3–5 years of age and adults.

Table 5-24. Geometric Mean and Selected Percentiles of Total Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2003–2004	0.797 (0.703–0.903)	0.800 (0.700–0.900)	1.70 (1.50–1.90)	3.30 (2.90–3.90)	4.90 (4.30–5.50)	8,373
	2005–2006	0.863 (0.787–0.946)	0.830 (0.760–0.920)	1.66 (1.48–1.93)	3.20 (2.87–3.54)	4.64 (4.17–5.25)	8,407
	2007–2008	0.769 (0.689–0.859)	0.740 (0.660–0.830)	1.48 (1.29–1.69)	2.95 (2.46–3.59)	4.64 (3.74–5.79)	8,266
	2009–2010	0.863 (0.792–0.941)	0.790 (0.730–0.880)	1.68 (1.49–1.91)	3.43 (3.07–3.84)	5.13 (4.57–5.67)	8,793
Age group							
1–5 years	2003–2004	0.326 (0.285–0.372)	0.300 (0.300–0.300)	0.500 (0.500–0.700)	1.00 (0.800–1.60)	1.80 (1.30–2.50)	911
	2005–2006	NC	<LOD	0.500 (0.470–0.550)	.940 (0.820–1.24)	1.43 (1.25–1.59)	968
	2007–2008	NC	<LOD	0.440 (0.380–0.540)	.830 (0.620–1.12)	1.32 (0.960–2.40)	817
	2009–2010	NC	<LOD	0.490 (0.430–0.590)	.890 (0.740–1.08)	1.30 (1.08–1.52)	836
6–11 years	2003–2004	0.419 (0.363–0.484)	0.400 (0.400–0.500)	0.700 (0.700–0.900)	1.30 (1.00–1.60)	1.90 (1.40–3.50)	856
	2005–2006	NC	0.410 (0.330–0.460)	0.740 (0.630–1.00)	1.43 (1.21–1.87)	2.34 (1.53–3.42)	934
	2007–2008	NC	0.380 (0.340–0.440)	0.700 (0.600–0.790)	1.21 (0.970–1.36)	1.56 (1.34–1.80)	1,011
	2009–2010	NC	0.360 (<LOD–0.400)	0.670 (0.590–0.770)	1.22 (1.05–1.45)	1.88 (1.43–2.61)	1,009
12–19 years	2003–2004	0.490 (0.418–0.574)	0.500 (0.400–0.600)	1.00 (0.800–1.20)	1.80 (1.40–2.30)	2.60 (2.10–3.30)	2,081
	2005–2006	0.513 (0.461–0.570)	0.460 (0.390–0.530)	0.850 (0.740–1.04)	1.66 (1.31–1.98)	2.41 (2.12–2.90)	1,996
	2007–2008	0.469 (0.426–0.516)	0.440 (0.390–0.490)	0.800 (0.670–0.970)	1.55 (1.30–1.72)	2.05 (1.77–2.34)	1,074
	2009–2010	0.534 (0.473–0.602)	0.450 (0.370–0.540)	0.910 (0.770–1.11)	2.04 (1.53–2.55)	3.01 (2.53–3.63)	1,183
≥20 years	2003–2004	0.979 (0.860–1.12)	1.00 (0.800–1.10)	2.00 (1.70–2.30)	3.80 (3.20–4.40)	5.40 (4.60–6.70)	4,525
	2005–2006	1.06 (0.967–1.15)	1.03 (0.930–1.15)	1.98 (1.73–2.22)	3.64 (3.33–4.01)	5.31 (4.82–5.67)	4,509
	2007–2008	0.944 (0.833–1.07)	0.890 (0.780–1.03)	1.73 (1.47–2.09)	3.41 (2.82–4.17)	5.32 (4.32–6.72)	5,364
	2009–2010	1.04 (0.956–1.14)	0.970 (0.870–1.08)	2.00 (1.80–2.20)	3.96 (3.55–4.27)	5.75 (5.14–6.50)	5,765
Gender							
Males	2003–2004	0.814 (0.714–0.927)	0.800 (0.700–0.900)	1.80 (1.50–2.00)	3.70 (3.20–4.30)	5.40 (4.60–6.50)	4,132
	2005–2006	0.864 (0.783–0.954)	0.810 (0.720–0.940)	1.69 (1.48–2.01)	3.30 (2.86–3.73)	4.83 (4.08–5.45)	4,092
	2007–2008	0.809 (0.709–0.923)	0.760 (0.670–0.850)	1.56 (1.31–1.81)	3.21 (2.72–4.06)	5.16 (4.12–6.97)	4,147
	2009–2010	0.883 (0.810–0.962)	0.790 (0.730–0.870)	1.75 (1.54–2.02)	3.84 (3.35–4.26)	5.65 (5.13–6.34)	4,366
Females	2003–2004	0.781 (0.689–0.886)	0.800 (0.700–0.900)	1.60 (1.40–1.80)	3.00 (2.50–3.50)	4.40 (3.60–5.30)	4,241
	2005–2006	0.864 (0.791–0.943)	0.850 (0.770–0.920)	1.63 (1.44–1.89)	3.09 (2.75–3.46)	4.51 (4.01–5.28)	4,315
	2007–2008	0.748 (0.677–0.827)	0.720 (0.660–0.810)	1.42 (1.24–1.60)	2.70 (2.27–3.27)	3.93 (3.17–5.16)	4,119
	2009–2010	0.845 (0.772–0.924)	0.800 (0.720–0.880)	1.61 (1.43–1.81)	3.13 (2.76–3.48)	4.43 (4.04–5.11)	4,427

Table 5-24. Geometric Mean and Selected Percentiles of Total Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Race/ethnicity							
Mexican Americans	2003–2004	0.563 (0.472–0.672)	0.600 (0.500–0.700)	1.00 (0.800–1.30)	1.90 (1.60–2.40)	3.00 (2.20–3.80)	2,085
	2005–2006	0.597 (0.524–0.679)	0.580 (0.490–0.670)	1.04 (0.870–1.24)	1.70 (1.40–2.12)	2.58 (1.96–3.31)	2,236
	2007–2008	0.594 (0.536–0.658)	0.580 (0.520–0.670)	1.03 (0.900–1.17)	1.73 (1.49–2.04)	2.48 (2.10–2.91)	1,712
	2009–2010	0.613 (0.571–0.659)	0.580 (0.540–0.630)	1.01 (0.890–1.15)	1.63 (1.47–1.90)	2.45 (2.03–2.93)	1,966
Non-Hispanic blacks	2003–2004	0.877 (0.753–1.02)	0.900 (0.800–1.00)	1.60 (1.40–1.80)	3.00 (2.30–4.00)	4.40 (3.30–6.00)	2,293
	2005–2006	0.823 (0.697–0.972)	0.800 (0.670–0.940)	1.50 (1.21–1.92)	2.72 (2.14–3.59)	4.09 (3.22–5.16)	2,193
	2007–2008	0.766 (0.711–0.825)	0.780 (0.710–0.830)	1.32 (1.23–1.42)	2.25 (1.99–2.58)	3.42 (2.74–3.90)	1,746
	2009–2010	0.928 (0.805–1.07)	0.900 (0.800–1.02)	1.67 (1.38–1.96)	2.93 (2.20–4.21)	4.56 (3.34–6.69)	1,593
Non-Hispanic whites	2003–2004	0.776 (0.655–0.919)	0.800 (0.700–0.900)	1.70 (1.40–2.00)	3.20 (2.60–3.90)	4.70 (4.00–5.60)	3,478
	2005–2006	0.891 (0.801–0.992)	0.870 (0.770–1.00)	1.74 (1.50–2.10)	3.37 (2.88–3.76)	4.76 (4.18–5.37)	3,310
	2007–2008	0.743 (0.651–0.847)	0.720 (0.620–0.820)	1.43 (1.18–1.70)	2.79 (2.33–3.41)	4.18 (3.57–4.83)	3,461
	2009–2010	0.856 (0.766–0.957)	0.790 (0.690–0.920)	1.70 (1.46–1.98)	3.43 (2.94–3.94)	4.92 (4.30–5.65)	3,760

[a] The limits of detection for survey years 2003–2004, 2005–2006, 2007–2008, and 2009–2010 were 0.2, 0.33, 0.33, and 0.33 µg/L, respectively.

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

Source: CDC 2024

Table 5-25. Geometric Mean and Selected Percentiles of Total Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2011–2012	0.703 (0.617–0.801)	0.640 (0.580–0.730)	1.38 (1.14–1.72)	2.87 (2.39–3.62)	4.40 (3.50–5.71)	7,920
	2013–2014	0.683 (0.621–0.751)	0.620 (0.540–0.690)	1.29 (1.14–1.46)	2.65 (2.32–3.08)	4.36 (3.65–4.97)	5,215
	2015–2016	0.678 (0.619–0.743)	0.600 (0.540–0.690)	1.26 (1.07–1.47)	2.55 (2.17–3.10)	4.25 (3.44–4.94)	4,988
	2017–2018	0.643 (0.577–0.715)	0.580 (0.490–0.690)	1.22 (1.08–1.38)	2.52 (2.17–2.91)	3.87 (3.41–4.61)	7,513
Age group							
1–5 years	2011–2012	0.262 (0.237–0.291)	0.250 (0.020–0.270)	0.390 (0.340–0.450)	0.680 (0.540–0.880)	0.99 (0.790–1.21)	713
	2013–2014	NC	<LOD	0.410 (0.370–0.450)	0.810 (0.710–0.990)	1.21 (1.05–1.48)	818
	2015–2016	NC	<LOD	0.380 (0.340–0.430)	0.690 (0.540–0.830)	1.06 (0.840–1.36)	790
	2017–2018	NC	<LOD	0.350 (0.310–0.420)	0.640 (0.520–0.790)	0.960 (0.750–1.22)	629
6–11 years	2011–2012	0.330 (0.287–0.379)	0.320 (0.280–0.360)	0.530 (0.0480–0.600)	0.930 (0.0780–1.20)	1.40 (1.02–2.17)	1,048
	2013–2014	NC	0.300 (<LOD–0.360)	0.570 (0.0470–0.680)	1.12 (0.0980–1.36)	1.62 (1.38–2.19)	1,075
	2015–2016	NC	0.310 (0.290–0.340)	0.480 (0.0430–0.570)	0.920 (0.0750–1.13)	1.33 (1.01–2.28)	1,023
	2017–2018	NC	<LOD	0.450 (0.380–0.550)	0.930 (0.710–1.11)	1.71 (1.02–2.41)	883
12–19 years	2011–2012	0.411 (0.355–0.476)	0.370 (0.320–0.450)	0.680 (0.0590–0.800)	1.32 (1.08–1.75)	2.25 (1.46–2.87)	1,129
	2013–2014	0.412 (0.367–0.463)	0.350 (0.310–0.420)	0.630 (0.0530–0.750)	1.20 (0.0900–1.67)	1.87 (1.30–2.38)	627
	2015–2016	0.395 (0.356–0.439)	0.340 (0.300–0.370)	0.590 (0.0470–0.750)	1.00 (0.0850–1.35)	1.89 (1.02–3.34)	565
	2017–2018	NC	0.310 (<LOD–0.370)	0.590 (0.500–0.700)	1.14 (0.890–1.47)	1.71 (1.31–2.73)	1,030
≥20 years	2011–2012	0.863 (0.753–0.990)	0.790 (0.690–0.940)	1.68 (1.36–2.12)	3.35 (2.71–4.31)	5.02 (3.94–6.96)	5,030
	2013–2014	0.814 (0.736–0.900)	0.740 (0.650–0.850)	1.54 (1.36–1.71)	3.08 (2.73–3.56)	4.88 (4.36–5.21)	2,695
	2015–2016	0.810 (0.740–0.886)	0.740 (0.660–0.830)	1.47 (1.28–1.75)	2.86 (2.50–3.44)	4.66 (3.91–5.96)	2,610
	2017–2018	0.764 (0.685–0.815)	0.730 (0.620–0.840)	1.40 (1.24–1.62)	3.00 (2.58–3.29)	4.36 (3.78–5.02)	5,021
Gender							
Males	2011–2012	0.712 (0.623–0.815)	0.650 (0.570–0.730)	1.40 (1.17–1.72)	3.00 (2.44–3.91)	4.94 (3.50–6.79)	3,968
	2013–2014	0.688 (0.617–0.767)	0.620 (0.530–0.720)	1.30 (1.12–1.54)	2.76 (2.36–3.34)	4.52 (3.65–5.23)	2,587
	2015–2016	0.679 (0.621–0.743)	0.610 (0.530–0.700)	1.29 (1.06–1.60)	2.45 (2.06–3.31)	4.67 (3.77–5.39)	2,488
	2017–2018	0.655 (0.577–0.743)	0.580 (0.470–0.730)	1.24 (1.05–1.52)	2.83 (2.41–3.18)	4.03 (3.53–4.56)	3,666
Females	2011–2012	0.694 (0.609–0.791)	0.640 (0.580–0.740)	1.36 (1.09–1.75)	2.81 (2.28–3.50)	4.03 (3.29–5.08)	3,952
	2013–2014	0.678 (0.617–0.745)	0.610 (0.530–0.700)	1.27 (1.14–1.42)	2.56 (2.17–3.08)	4.15 (3.37–4.93)	2,628
	2015–2016	0.677 (0.608–0.754)	0.600 (0.530–0.700)	1.23 (1.02–1.42)	2.57 (2.20–3.10)	3.95 (3.10–4.55)	2,500
	2017–2018	0.631 (0.573–0.695)	0.580 (0.500–0.560)	1.19 (1.08–1.35)	2.23 (1.96–2.69)	3.78 (3.13–4.68)	3,847

Table 5-25. Geometric Mean and Selected Percentiles of Total Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Race/ethnicity							
Mexican Americans	2011–2012	0.483 (0.424–0.550)	0.480 (0.400–0.560)	0.810 (0.720–0.900)	1.44 (1.16–1.63)	1.90 (1.57–2.19)	1,077
	2013–2014	0.487 (0.433–0.547)	0.430 (0.390–0.510)	0.760 (0.690–0.870)	1.41 (1.14–1.69)	1.98 (1.70–2.38)	969
	2015–2016	0.540 (0.522–0.559)	0.530 (0.470–0.570)	0.840 (0.790–0.920)	1.41 (1.19–1.60)	1.81 (1.54–2.33)	994
	2017–2018	0.513 (0.444–0.593)	0.430 (0.370–0.500)	0.910 (0.680–1.15)	1.68 (1.38–2.11)	2.92 (1.98–4.41)	1,134
Non-Hispanic blacks	2011–2012	0.679 (0.542–0.852)	0.630 (0.500–0.790)	1.24 (0.0880–1.72)	2.45 (1.84–3.14)	3.80 (2.70–5.37)	2,195
	2013–2014	0.699 (0.614–0.796)	0.650 (0.570–0.750)	1.20 (1.08–1.40)	2.30 (1.65–2.96)	3.34 (2.35–5.93)	1,119
	2015–2016	0.698 (0.587–0.829)	0.630 (0.510–0.760)	1.21 (1.04–1.47)	2.49 (2.01–3.51)	4.51 (2.74–5.87)	1,070
	2017–2018	0.655 (0.558–0.769)	0.580 (0.460–0.730)	1.20 (0.980–1.51)	2.54 (1.93–3.38)	4.35 (3.40–5.09)	1,708
Non-Hispanic whites	2011–2012	0.688 (0.582–0.813)	0.630 (0.550–0.750)	1.38 (1.09–1.82)	2.83 (2.18–3.82)	4.25 (3.02–6.24)	2,493
	2013–2014	0.672 (0.598–0.755)	0.620 (0.520–0.720)	1.30 (1.12–1.51)	2.61 (2.18–3.08)	4.15 (3.35–4.98)	1,848
	2015–2016	0.638 (0.563–0.723)	0.570 (0.500–0.670)	1.22 (0.940–1.47)	2.29 (1.82–2.72)	3.95 (2.56–4.55)	1,511
	2017–2018	0.623 (0.546–0.711)	0.580 (0.470–0.730)	1.19 (1.03–1.35)	2.23 (1.93–2.64)	3.46 (2.89–4.13)	2,536
All Hispanics	2011–2012	0.612 (0.527–0.710)	0.590 (0.490–0.700)	1.08 (0.890–1.33)	1.96 (1.60–2.68)	3.03 (2.37–3.86)	1,931
	2013–2014	0.551 (0.486–0.624)	0.490 (0.420–0.580)	0.910 (0.820–1.10)	1.76 (1.44–2.12)	2.59 (2.06–3.14)	1,481
	2015–2016	0.607 (0.546–0.675)	0.570 (0.520–0.640)	1.00 (0.870–1.17)	1.75 (1.38–2.33)	2.60 (1.85–3.25)	1,664
	2017–2018	0.591 (0.522–0.669)	0.520 (0.430–0.620)	1.07 (0.910–1.27)	2.04 (1.72–2.58)	3.23 (2.65–4.41)	1,816
Asians	2011–2012	1.86 (1.58–2.19)	2.30 (1.84–2.64)	4.32 (3.71–5.21)	7.71 (6.38–8.79)	10.3 (8.85–12.0)	1,005
	2013–2014	1.72 (1.46–2.03)	1.77 (1.42–2.26)	3.92 (3.35–4.55)	7.78 (6.39–9.16)	9.99 (9.16–13.7)	510
	2015–2016	1.73 (1.41–2.12)	2.03 (1.40–2.70)	4.21 (3.31–5.50)	7.66 (6.17–9.91)	11.3 (9.12–13.8)	479
	2017–2018	1.24 (0.964–1.60)	1.28 (0.930–1.73)	3.28 (2.49–4.01)	5.87 (4.83–8.44)	9.51 (6.97–12.2)	946

[a] The limits of detection for survey years 2011–2012, 2013–2014, 2015–2016, and 2017–2018 were 0.16, 0.28, 0.28, and 0.28 µg/L, respectively.

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

Source: CDC 2024

Table 5-26. Geometric Mean and Selected Percentiles of Inorganic Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

			Selected percentiles (95% CI)[a]				
	Survey years	Geometric mean (95% CI)	50th	75th	90th	95th	Sample size
Total	2003–2004	NC	<LOD	<LOD	0.600 (0.500–0.600)	0.700 (0.700–0.700)	8,147
	2005–2006	NC	<LOD	<LOD	0.540 (0.500–0.580)	0.660 (0.620–0.710)	8,371
	2007–2008	NC	<LOD	0.350 (<LOD–0.370)	0.520 (0.500–0.540)	0.650 (0.620–0.690)	8,162
	2009–2010	NC	<LOD	<LOD	0.390 (0.360–0.430)	0.510 (0.480–0.570)	8,733
Age group							
1–5 years	2003–2004	NC	<LOD	<LOD	<LOD	0.500 (<LOD–0.600)	792
	2005–2006	NC	<LOD	<LOD	0.430 (<LOD–0.470)	0.510 (0.430–0.670)	948
	2007–2008	NC	<LOD	<LOD	0.350 (<LOD–0.450)	0.500 (0.410–0.550)	726
	2009–2010	NC	<LOD	<LOD	<LOD	0.360 (<LOD–0.460)	789
6–11 years	2003–2004	NC	<LOD	<LOD	<LOD	0.600 (0.500–0.600)	842
	2005–2006	NC	<LOD	<LOD	0.450 (<LOD–0.520)	0.560 (0.470–0.640)	932
	2007–2008	NC	<LOD	<LOD	0.380 (0.350–0.410)	0.470 (0.420–0.520)	1,010
	2009–2010	NC	<LOD	<LOD	<LOD	0.380 (0.350–0.440)	1,006
12–19 years	2003–2004	NC	<LOD	<LOD	0.500 (<LOD–0.500)	0.600 (0.500–0.600)	2,060
	2005–2006	NC	<LOD	<LOD	0.430 (0.410–0.460)	0.540 (0.480–0.590)	1,984
	2007–2008	NC	<LOD	<LOD	0.370 (<LOD–0.400)	0.480 (0.410–0.530)	1,069
	2009–2010	NC	<LOD	<LOD	<LOD	0.420 (0.350–0.500)	1,184
≥20 years	2003–2004	NC	<LOD	<LOD	0.600 (0.500–0.600)	0.700 (0.700–0.800)	4,453
	2005–2006	NC	<LOD	<LOD	0.570 (0.530–0.610)	0.690 (0.650–0.750)	4,507
	2007–2008	NC	<LOD	0.380 (0.360–0.390)	0.550 (0.530–0.570)	0.700 (0.660–0.730)	5,357
	2009–2010	NC	<LOD	<LOD	0.420 (0.390–0.450)	0.540 (0.490–0.600)	5,754
Gender							
Males	2003–2004	NC	<LOD	<LOD	0.500 (0.500–0.600)	0.600 (0.600–0.700)	4,015
	2005–2006	NC	<LOD	<LOD	0.480 (0.450–0.520)	0.600 (0.550–0.640)	4,076
	2007–2008	NC	<LOD	<LOD	0.500 (0.470–0.520)	0.600 (0.570–0.650)	4,093
	2009–2010	NC	<LOD	<LOD	0.370 (<LOD–0.420)	0.500 (0.440–0.560)	4,336
Females	2003–2004	NC	<LOD	<LOD	0.600 (0.500–0.600)	0.700 (0.700–0.800)	4,132
	2005–2006	NC	<LOD	<LOD	0.580 (0.550–0.640)	0.700 (0.670–0.780)	4,295
	2007–2008	NC	<LOD	0.380 (0.360–0.390)	0.550 (0.520–0.570)	0.700 (0.670–0.740)	4,069
	2009–2010	NC	<LOD	<LOD	0.410 (0.380–0.440)	0.530 (0.490–0.600)	4,397

Table 5-26. Geometric Mean and Selected Percentiles of Inorganic Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a] 50th	75th	90th	95th	Sample size
Race/ethnicity							
Mexican Americans	2003–2004	NC	<LOD	<LOD	0.500 (0.500–0.600)	0.700 (0.600–0.800)	2,007
	2005–2006	NC	<LOD	<LOD	0.530 (0.470–0.580)	0.670 (0.560–0.830)	2,224
	2007–2008	NC	<LOD	<LOD	0.430 (0.400–0.480)	0.560 (0.520–0.610)	1,685
	2009–2010	NC	<LOD	<LOD	<LOD	0.470 (0.390–0.530)	1,947
Non-Hispanic blacks	2003–2004	NC	<LOD	<LOD	0.600 (0.500–0.600)	0.700 (0.600–0.800)	2,240
	2005–2006	NC	<LOD	<LOD	0.530 (0.470–0.600)	0.670 (0.600–0.760)	2,183
	2007–2008	NC	<LOD	<LOD	0.490 (0.450–0.530)	0.610 (0.560–0.650)	1,729
	2009–2010	NC	<LOD	<LOD	0.370 (0.350–0.390)	0.480 (0.410–0.530)	1,580
Non-Hispanic whites	2003–2004	NC	<LOD	<LOD	0.600 (0.500–0.600)	0.700 (0.600–0.700)	3,406
	2005–2006	NC	<LOD	<LOD	0.540 (0.500–0.580)	0.650 (0.610–0.710)	3,298
	2007–2008	NC	<LOD	0.360 (<LOD–0.390)	0.530 (0.500–0.550)	0.660 (0.620–0.700)	3,421
	2009–2010	NC	<LOD	<LOD	0.410 (0.370–0.450)	0.520 (0.480–0.590)	3,739

[a] The limits of detection for survey years 2003–2004, 2005–2006, 2007–2008, and 2009–2010 were 0.42, 0.4, 0.35, and 0.35 µg/L, respectively.

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

Source: CDC 2024

Table 5-27. Geometric Mean and Selected Percentiles of Inorganic Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

		Geometric mean	Selected percentiles (95% CI)[a]				Sample
	Survey years	(95% CI)	50th	75th	90th	95th	size
Total	2011–2012	NC	<LOD	<LOD	0.440 (0.390–0.480)	0.600 (0.520–0.680)	7,841
	2013–2014	NC	<LOD	<LOD	0.410 (0.380–0.440)	0.530 (0.490–0.570)	5,175
	2015–2016	NC	<LOD	<LOD	0.350 (0.300–0.400)	0.480 (0.430–0.530)	4,938
	2017–2018	NC	<LOD	<LOD	0.320 (0.300–0.350)	0.450 (0.400–0.510)	7,480
Age group							
1–5 years	2011–2012	NC	<LOD	<LOD	<LOD	0.280 (<LOD–0.360)	657
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	779
	2015–2016	NC	<LOD	<LOD	<LOD	0.270 (<LOD–0.340)	749
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	611
6–11 years	2011–2012	NC	<LOD	<LOD	0.280 (<LOD–0.320)	0.360 (0.290–0.450)	1,044
	2013–2014	NC	<LOD	<LOD	<LOD	0.340 (<LOD–0.420)	1,074
	2015–2016	NC	<LOD	<LOD	<LOD	0.330 (<LOD–0.450)	1,022
	2017–2018	NC	<LOD	<LOD	<LOD	0.280 (0.230–0.390)	831
12–19 years	2011–2012	NC	<LOD	<LOD	0.280 (<LOD–0.350)	0.400 (0.300–0.540)	1,121
	2013–2014	NC	<LOD	<LOD	<LOD	0.360 (0.290–0.420)	627
	2015–2016	NC	<LOD	<LOD	<LOD	0.400 (<LOD–0.510)	565
	2017–2018	NC	<LOD	<LOD	<LOD	0.280 (0.240–0.350)	1,027
≥20 years	2011–2012	NC	<LOD	0.290 (0.270–0.310)	0.470 (0.420–0.530)	0.630 (0.550–0.760)	5,019
	2013–2014	NC	<LOD	0.270 (<LOD–0.300)	0.440 (0.410–0.470)	0.560 (0.510–0.660)	2,695
	2015–2016	NC	<LOD	<LOD	0.380 (0.330–0.430)	0.500 (0.450–0.560)	2,602
	2017–2018	NC	<LOD	0.210 (<LOD–0.230)	0.350 (0.320–0.370)	0.500 (0.420–0.560)	5,011
Gender							
Males	2011–2012	NC	<LOD	<LOD	0.420 (0.370–0.480)	0.600 (0.490–0.680)	3,925
	2013–2014	NC	<LOD	<LOD	0.400 (0.340–0.430)	0.510 (0.450–0.560)	2,570
	2015–2016	NC	<LOD	<LOD	0.330 (0.280–0.400)	0.430 (0.400–0.490)	2,460
	2017–2018	NC	<LOD	<LOD	0.310 (0.280–0.330)	0.430 (0.380–0.480)	3,650
Females	2011–2012	NC	<LOD	0.280 (<LOD–0.300)	0.450 (0.390–0.490)	0.610 (0.520–0.700)	3,916
	2013–2014	NC	<LOD	<LOD	0.420 (0.390–0.450)	0.550 (0.510–0.590)	2,605
	2015–2016	NC	<LOD	<LOD	0.370 (0.320–0.450)	0.530 (0.450–0.620)	2,478
	2017–2018	NC	<LOD	<LOD	0.330 (0.300–0.370)	0.480 (0.400–0.550)	3,380

Table 5-27. Geometric Mean and Selected Percentiles of Inorganic Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

			Selected percentiles (95% CI)[a]				
	Survey years	Geometric mean (95% CI)	50th	75th	90th	95th	Sample size
Race/ethnicity							
Mexican Americans	2011–2012	NC	<LOD	<LOD	0.390 (0.330–0.460)	0.580 (0.460–0.710)	1,058
	2013–2014	NC	<LOD	<LOD	0.380 (0.330–0.410)	0.540 (0.420–0.640)	958
	2015–2016	NC	<LOD	<LOD	0.420 (0.290–0.600)	0.560 (0.430–0.820)	988
	2017–2018	NC	<LOD	<LOD	0.370 (0.290–0.490)	0.590 (0.450–0.800)	1,129
Non-Hispanic blacks	2011–2012	NC	<LOD	<LOD	0.410 (0.370–0.470)	0.570 (0.480–0.670)	2,170
	2013–2014	NC	<LOD	<LOD	0.380 (0.330–0.430)	0.530 (0.440–0.630)	1,110
	2015–2016	NC	<LOD	<LOD	0.330 (0.270–0.380)	0.430 (0.370–0.490)	1,058
	2017–2018	NC	<LOD	<LOD	0.320 (0.280–0.370)	0.430 (0.380–0.520)	1,129
Non-Hispanic whites	2011–2012	NC	<LOD	0.270 (<LOD–0.300)	0.430 (0.370–0.490)	0.590 (0.480–0.690)	2,477
	2013–2014	NC	<LOD	<LOD	0.420 (0.360–0.450)	0.510 (0.460–0.570)	1,835
	2015–2016	NC	<LOD	<LOD	0.320 (0.280–0.400)	0.460 (0.400–0.540)	1,500
	2017–2018	NC	<LOD	<LOD	0.300 (0.260–0.360)	0.410 (0.360–0.500)	2,525
All Hispanics	2011–2012	NC	<LOD	<LOD	0.430 (0.370–0.490)	0.630 (0.540–0.760)	1,902
	2013–2014	NC	<LOD	<LOD	0.380 (0.330–0.410)	0.540 (0.440–0.650)	1,467
	2015–2016	NC	<LOD	<LOD	0.380 (0.300–0.490)	0.510 (0.430–0.650)	1,648
	2017–2018	NC	<LOD	<LOD	0.360 (0.310–0.430)	0.540 (0.450–0.650)	1,810
Asians	2011–2012	NC	<LOD	0.350 (0.310–0.390)	0.550 (0.500–0.590)	0.700 (0.630–0.760)	997
	2013–2014	NC	<LOD	0.330 (0.270–0.390)	0.570 (0.420–0.680)	0.750 (0.580–1.10)	508
	2015–2016	NC	<LOD	<LOD	0.430 (0.360–0.540)	0.600 (0.470–0.730)	473
	2017–2018	NC	<LOD	0.240 (<LOD–0.310)	0.420 (0.310–0.540)	0.550 (0.420–0.800)	944

[a] The limits of detection for survey years 2011–2012, 2013–2014, 2015–2016, and 2017–2018 were 0.27, 0.27, 0.27, and 0.27 µg/L, respectively.

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

Source: CDC 2024

Table 5-28. Geometric Mean and Selected Percentiles of Ethyl Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	7,841
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	5,175
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	4,936
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	7,480
Age group							
1–5 years	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	657
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	779
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	749
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	611
6–11 years	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	1,044
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	1,074
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	1,022
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	831
12–19 years	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	1,121
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	627
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	565
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	1,027
≥20 years	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	5,019
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	2,695
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	2,600
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	5,011
Gender							
Males	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	3,925
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	2,570
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	2,458
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	3,650
Females	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	3,916
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	2,605
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	2,478
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	3,830

Table 5-28. Geometric Mean and Selected Percentiles of Ethyl Mercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a] 50th	75th	90th	95th	Sample size
Race/ethnicity							
Mexican Americans	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	1,058
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	958
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	988
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	1,129
Non-Hispanic blacks	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	2,170
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	1,110
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	1,057
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	1,699
Non-Hispanic whites	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	2,477
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	1,835
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	1,500
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	2,525
All Hispanics	2011–2012	NC	<LOD	<LOD	<LOD	<LOD	1,902
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	1,467
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	1,647
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	1,180
Asians	2011–2012	NC	<LOD	<LOD	<LOD	0.160 (<LOD–0.280)	997
	2013–2014	NC	<LOD	<LOD	<LOD	<LOD	508
	2015–2016	NC	<LOD	<LOD	<LOD	<LOD	473
	2017–2018	NC	<LOD	<LOD	<LOD	<LOD	944

[a] The limits of detection for survey years 2011–2012, 2013–2014, 2015–2016, and 2017–2018 were 0.16, 0.16, 0.16, and 0.064 µg/L, respectively.
.

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

Source: CDC 2024

Table 5-29. Geometric Mean and Selected Percentiles of Methylmercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2011–2012	0.498 (0.423–0.587)	0.480 (0.400–0.570)	1.25 (0.950–1.61)	2.81 (2.29–3.55)	4.43 (3.46–5.49)	7,841
	2013–2014	0.434 (0.381–0.495)	0.420 (0.340–0.510)	1.09 (0.940–1.27)	2.62 (2.18–3.04)	4.28 (3.74–4.93)	5,175
	2015–2016	0.413 (0.361–0.472)	0.380 (0.320–0.490)	1.02 (0.860–1.22)	2.30 (1.92–2.78)	3.92 (3.35–4.81)	4,938
	2017–2018	0.480 (0.431–0535)	0.390 (0.210–0.470)	0.980 (0.830–1.15)	2.23 (1.88–2.57)	3.49 (3.05–4.01)	7,480
Age group							
1–5 years	2011–2012	NC	0.140 (0.120–0.170)	0.270 (0.220–0.350)	0.540 (0.420–0.780)	0.970 (0.590–1.14)	657
	2013–2014	NC	<LOD	0.260 (0.230–0.310)	0.660 (0.550–0.810)	1.11 (0.960–1.48)	779
	2015–2016	NC	<LOD	0.200 (0.170–0.240)	0.470 (0.350–0.630)	0.830 (0.590–1.21)	749
	2017–2018	NC	<LOD	<LOD	0.460 (0.380–0.600)	0.790 (0.600–0.930)	611
6–11 years	2011–2012	0.209 (0.182–0.241)	0.180 (0.150–0.220)	0.400 (0.330–0.490)	0.820 (0.630–1.06)	1.34 (0.940–1.84)	1,044
	2013–2014	NC	0.150 (<LOD–0.200)	0.380 (0.280–0.530)	0.960 (0.700–1.26)	1.58 (1.26–2.11)	1,074
	2015–2016	NC	0.140 (<LOD–0.180)	0.290 (0.260–0.340)	0.730 (0.530–0.950)	1.11 (0.870–1.91)	1,022
	2017–2018	NC	<LOD	0.280 (<LOD–0.340)	0.710 (0.490–0.960)	1.40 (0.830–2.02)	831
12–19 years	2011–2012	0.276 (0.237–0.322)	0.270 (0.210–0.310)	0.570 (0.460–0.670)	1.27 (0.870–1.67)	2.15 (1.40–2.81)	1121
	2013–2014	0.233 (0.202–0.269)	0.190 (0.140–0.260)	0.480 (0.380–0.560)	1.02 (0.730–1.60)	1.84 (1.20–2.57)	627
	2015–2016	0.212 (0.180–0.249)	0.160 (0.120–0.220)	0.420 (0.310–0.620)	0.800 (0.690–1.23)	1.81 (0.800–3.14)	565
	2017–2018	NC	<LOD	0.410 (0.340–0.510)	0.990 (0.730–1.27)	1.52 (1.05–2.41)	1,027
≥20 years	2011–2012	0.624 (0.523–0.746)	0.610 (0.500–0.760)	1.53 (1.18–2.00)	3.28 (2.56–4.31)	4.97 (3.91–6.89)	5,019
	2013–2014	0.541 (0.473–0.618)	0.540 (0.450–0.630)	1.32 (1.17–1.54)	3.05 (2.64–3.60)	4.92 (4.34–5.41)	2,695
	2015–2016	0.518 (0.456–0.588)	0.530 (0.420–0.630)	1.23 (1.05–1.46)	2.59 (2.25–3.36)	4.42 (3.66–5.58)	2,602
	2017–2018	0.566 (0.505–0.635)	0.500 (0.420–0.610)	1.17 (1.01–1.34)	2.61 (2.23–2.93)	3.89 (3.31–4.53)	5,011
Gender							
Males	2011–2012	0.509 (0.433–0.598)	0.490 (0.400–0.590)	1.30 (0.990–1.62)	2.84 (2.29–3.68)	4.77 (3.44–6.74)	3,925
	2013–2014	0.448 (0.390–0.515)	0.430 (0.350–0.530)	1.10 (0.940–1.37)	2.67 (2.22–3.33)	4.44 (3.88–5.40)	2,570
	2015–2016	0.429 (0.379–0.486)	0.400 (0.330–0.520)	1.11 (0.900–1.33)	2.24 (1.85–2.86)	4.42 (3.60–5.05)	2,460
	2017–2018	0.496 (0.434–0.567)	0.390 (0.300–0.510)	1.04 (0.820–1.28)	2.51 (2.07–2.78)	3.69 (3.19–4.13)	3,650
Females	2011–2012	0.489 (0.413–0.580)	0.470 (0.380–0.560)	1.19 (0.900–1.61)	2.72 (2.18–3.46)	3.99 (3.28–4.99)	3,916
	2013–2014	0.422 (0.367–0.485)	0.400 (0.330–0.490)	1.08 (0.900–1.22)	2.46 (2.03–2.92)	3.91 (3.32–4.93)	2,605
	2015–2016	0.399 (0.338–0.470)	0.370 (0.300–0.470)	0.950 (0.760–1.18)	2.36 (1.91–2.81)	3.58 (2.92–4.24)	2,478
	2017–2018	0.465 (0.424–0.511)	0.370 (0.310–0.450)	0.960 (0.810–1.10)	1.92 (1.64–2.43)	3.32 (2.77–3.90)	3,830

Table 5-29. Geometric Mean and Selected Percentiles of Methylmercury Blood Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Race/ethnicity							
Mexican Americans	2011–2012	0.320 (0.264–0.387)	0.330 (0.260–0.410)	0.610 (0.500–0.770)	1.23 (0.920–1.40)	1.66 (1.33–2.06)	1,058
	2013–2014	0.276 (0.239–0.318)	0.260 (0.210–0.310)	0.540 (0.440–0.650)	1.06 (0.770–1.43)	1.72 (1.27–2.34)	958
	2015–2016	0.292 (0.276–0.309)	0.290 (0.260–0.330)	0.610 (0.550–0.690)	1.13 (0.870–1.47)	1.54 (1.34–2.02)	988
	2017–2018	NC	0.270 (<LOD–0.320)	0.620 (0.460–0.820)	1.33 (0.990–1.87)	2.24 (1.51–3.08	1,129
Non-Hispanic blacks	2011–2012	0.517 (0.392–0.681)	0.510 (0.380–0.660)	1.13 (0.750–1.61)	2.37 (1.66–3.08)	3.63 (2.57–5.16)	2,170
	2013–2014	0.481 (0.407–0.569)	0.460 (0.370–0.580)	1.03 (0.900–1.22)	2.08 (1.55–2.72)	3.33 (2.15–5.12)	1,110
	2015–2016	0.461 (0.376–0.565)	0.430 (0.330–0.580)	1.05 (0.830–1.25)	2.34 (1.71–3.31)	4.08 (2.80–5.58)	1,058
	2017–2018	0.492 (0.412–0.589)	0.410 (0.290–0.540)	0.970 (0.740–1.28)	2.31 (1.68–2.94)	3.69 (3.04–4.45)	1,699
Non-Hispanic whites	2011–2012	0.478 (0.392–0.583)	0.470 (0.360–0.580)	1.25 (0.870–1.69)	2.76 (2.06–3.69)	4.24 (2.92–6.38)	2,477
	2013–2014	0.420 (0.357–0.495)	0.420 (0.310–0.530)	1.10 (0.910–1.32)	2.62 (2.07–3.12)	4.01 (3.46–5.00)	1,835
	2015–2016	0.380 (0.319–0.451)	0.350 (0.270–0.490)	0.960 (0.750–1.22)	2.04 (1.55–2.51)	3.50 (2.32–4.49)	1,500
	2017–2018	NC	0.380 (0.300–0.470)	0.950 (0.800–1.12)	1.91 (1.62–2.46)	3.19 (2.56–3.75)	2,525
All Hispanics	2011–2012	0.429 (0.350–0.525)	0.420 (0.340–0.520)	0.890 (0.700–1.17)	1.81 (1.39–2.46)	2.94 (2.19–3.71)	1,902
	2013–2014	0.326 (0.278–0.382)	0.310 (0.250–0.350)	0.680 (0.580–0.790)	1.50 (1.15–1.89)	2.34 (1.79–3.00)	1,467
	2015–2016	0.353 (0.305–0.409)	0.350 (0.300–0.390)	0.760 (0.630–0.930)	1.53 (1.14–1.98)	2.26 (1.60–3.03)	1,648
	2017–2018	NC	0.330 (0.270–0.410)	0.810 (0.670–0.990)	1.65 (1.32–2.23)	2.68 (2.08–3.86)	1,810
Asians	2011–2012	1.58 (1.29–1.94)	2.16 (1.68–2.55)	4.35 (3.64–5.13)	7.57 (6.21–8.61)	10.5 (8.48–12.5)	997
	2013–2014	1.42 (1.16–1.74)	1.71 (1.17–2.07)	3.90 (3.31–4.54)	7.93 (6.42–9.21)	10.8 (9.57–13.6)	508
	2015–2016	1.37 (1.08–1.73)	1.85 (1.18–2.51)	3.80 (3.11–4.83)	7.52 (5.60–9.63)	11.0 (8.65–12.4)	473
	2017–2018	0.996 (0.791–1.25)	1.07 (0.760–1.47)	2.90 (2.13–3.57)	5.35 (4.49–7.33)	8.67 (6.45–11.0)	994

[a] The limits of detection for survey years 2011–2012, 2013–2014, 2015–2016, and 2018–2018 were 0.12, 0.12, 0.12, and 0.26 µg/L, respectively.

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

Source: CDC 2024

Table 5-30. Geometric Mean and Selected Percentiles of Total Mercury Urinary Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2003–2004	0.447 (0.406–0.492)	0.420 (0.360–0.480)	1.00 (0.870–1.14)	2.08 (1.78–2.42)	3.19 (2.76–3.55)	2,538
	2005–2006	0.468 (0.426–0.514)	0.460 (0.410–0.510)	1.03 (0.900–1.12)	2.11 (1.88–2.36)	2.94 (2.58–3.26)	2,578
	2007–2008	0.443 (0.408–0.482)	0.440 (0.400–0.470)	0.880 (0.760–1.00)	1.74 (1.62–1.96)	2.66 (2.29–3.08)	2,634
	2009–2010	NC	0.400 (0.360–0.450)	0.850 (0.770–0.910)	1.53 (1.30–1.81)	2.42 (2.07–2.72)	2,865
Age group							
6–11 years	2003–2004	0.254 (0.213–0.304)	0.200 (0.160–0.250)	0.440 (0.330–0.580)	1.16 (0.610–1.61)	1.96 (1.13–2.97)	287
	2005–2006	0.333 (0.267–0.416)	0.320 (0.250–0.390)	0.650 (0.470–0.840)	1.32 (0.930–1.88)	2.18 (1.28–3.40)	355
	2007–2008	0.301 (0.260–0.347)	0.290 (0.230–0.340)	0.520 (0.430–0.620)	1.03 (0.770–1.23)	1.87 (1.03–3.48)	398
	2009–2010	NC	0.260 (0.220–0.320)	0.510 (0.430–0.620)	1.03 (0.730–1.31)	1.58 (1.18–1.88)	379
12–19 years	2003–2004	0.358 (0.313–0.408)	0.330 (0.290–0.370)	0.700 (0.530–0.840)	1.60 (1.14–2.52)	2.93 (1.88–3.66)	722
	2005–2006	0.372 (0.286–0.486)	0.350 (0.270–0.470)	0.740 (0.580–0.920)	1.61 (0.970–2.81)	2.59 (1.40–4.45)	703
	2007–2008	0.364 (0.326–0.406)	0.380 (0.320–0.450)	0.590 (0.550–0.650)	1.24 (0.830–1.71)	1.82 (1.41–2.29)	375
	2009–2010	NC	0.290 (0.230–0.360)	0.530 (0.470–0.630)	1.09 (0.890–1.31)	1.73 (1.28–2.31)	455
≥20 years	2003–2004	0.495 (0.442–0.555)	0.480 (0.410–0.570)	1.12 (0.930–1.29)	2.20 (1.85–2.65)	3.33 (2.76–3.88)	1,529
	2005–2006	0.505 (0.468–0.545)	0.510 (0.460–0.560)	1.11 (1.04–1.16)	2.23 (1.97–2.50)	3.11 (2.64–3.37)	1,520
	2007–2008	0.477 (0.435–0.523)	0.470 (0.430–0.520)	0.970 (0.850–1.10)	1.89 (1.69–2.20)	2.82 (2.33–3.56)	1,861
	2009–2010	NC	0.450 (0.390–0.510)	0.890 (0.810–1.00)	1.66 (1.40–2.01)	2.53 (2.21–2.84)	2,031
Gender							
Males	2003–2004	0.433 (0.405–0.463)	0.400 (0.350–0.460)	0.940 (0.840–1.05)	1.88 (1.63–2.18)	2.68 (2.34–3.05)	1,266
	2005–2006	0.464 (0.411–0.523)	0.450 (0.400–0.520)	0.980 (0.860–1.11)	2.03 (1.57–2.48)	3.00 (2.48–3.37)	1,270
	2007–2008	0.457 (0.417–0.501)	0.460 (0.400–0.520)	0.880 (0.780–1.01)	1.68 (1.53–1.77)	2.40 (2.11–2.76)	1,326
	2009–2010	NC	0.410 (0.340–0.480)	0.860 (0.750–0.950)	1.46 (1.29–1.66)	2.21 (1.93–2.53)	1,404
Females	2003–2004	0.460 (0.396–0.534)	0.430 (0.330–0.530)	1.07 (0.870–1.28)	2.26 (1.77–2.90)	3.54 (2.76–4.31)	1,272
	2005–2006	0.472 (0.424–0.525)	0.470 (0.390–0.550)	1.07 (0.900–1.19)	2.14 (1.84–2.50)	2.89 (2.60–3.38)	1,308
	2007–2008	0.431 (0.388–0.478)	0.430 (0.380–0.460)	0.870 (0.710–1.05)	1.88 (1.55–2.38)	2.92 (2.27–4.17)	1,308
	2009–2010	NC	0.390 (0.360–0.450)	0.840 (0.730–0.940)	1.61 (1.29–2.03)	2.61 (2.16–3.12)	1,461
Race/ethnicity							
Mexican Americans	2003–2004	0.416 (0.340–0.509)	0.360 (0.280–0.430)	0.960 (0.700–1.23)	2.19 (1.39–3.24)	3.16 (1.99–6.30)	619
	2005–2006	0.451 (0.369–0.551)	0.420 (0.310–0.560)	1.01 (0.780–1.25)	2.22 (1.48–2.64)	3.00 (2.27–4.01)	651
	2007–2008	0.409 (0.349–0.480)	0.370 (0.330–0.450)	0.780 (0.700–0.950)	1.82 (1.26–1.97)	2.55 (1.87–3.08)	514
	2009–2010	NC	0.350 (0.280–0.430)	0.670 (0.520–0.890)	1.53 (1.06–1.84)	2.29 (1.81–2.76)	615

Table 5-30. Geometric Mean and Selected Percentiles of Total Mercury Urinary Concentrations (in μg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Non-Hispanic blacks	2003–2004	0.476 (0.413–0.549)	0.430 (0.360–0.530)	0.890 (0.770–1.00)	1.96 (1.60–2.31)	3.09 (2.03–4.89)	713
	2005–2006	0.453 (0.384–0.533)	0.450 (0.380–0.550)	0.890 (0.710–1.13)	1.78 (1.34–2.29)	2.57 (2.21–3.15)	691
	2007–2008	0.478 (0.411–0.556)	0.460 (0.380–0.540)	0.910 (0.770–1.06)	1.85 (1.42–2.41)	2.76 (1.97–4.19)	589
	2009–2010	NC	0.410 (0.340–0.490)	0.840 (0.650–1.08)	1.66 (1.34–1.95)	2.64 (1.88–3.30)	546
Non-Hispanic whites	2003–2004	0.441 (0.382–0.509)	0.420 (0.330–0.520)	1.01 (0.840–1.23)	2.08 (1.67–2.46)	3.24 (2.67–3.60)	1,066
	2005–2006	0.459 (0.409–0.513)	0.440 (0.400–0.510)	1.00 (0.860–1.12)	2.07 (1.77–2.40)	2.81 (2.47–3.37)	1,044
	2007–2008	0.431 (0.378–0.493)	0.430 (0.380–0.480)	0.880 (0.700–1.07)	1.71 (1.50–2.18)	2.70 (2.18–3.59)	1,100
	2009–2010	NC	0.390 (0.330–0.470)	0.850 (0.750–0.950)	1.52 (1.26–2.01)	2.42 (1.93–2.85)	1,225

[a] The limits of detection for survey years 2003–2004, 2005–2006, 2007–2008, and 2009–2010 were 0.14, 0.11, 0.08, and 0.08 μg/L, respectively.

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

Source: CDC 2024

Table 5-31. Geometric Mean and Selected Percentiles of Total Mercury Urinary Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2011–2012	0.324 (0.285–0.368)	0.320 (0.280–0.370)	0.660 (0.580–0.770)	1.37 (1.15–1.59)	1.83 (1.62–2.14)	2,507
	2013–2014	0.246 (0.221–0.273)	0.200 (0.170–0.240)	0.470 (0.400–0.570)	1.07 (0.900–1.22)	1.64 (1.35–1.96)	2,666
	2015–2016	NC	<LOD	0.280 (0.250–0.320)	0.680 (0.570–0.780)	1.18 (0.920–1.29)	3,080
	2017–2018	NC	<LOD	0.300 (0.250–0.379)	0.680 (0.600–0.770)	1.05 (0.850–1.23)	2,812
Age group							
3–5 years	2015–2016	NC	<LOD	<LOD	0.160 (<LOD–0.240)	0.280 (0.190–0.510)	496
	2017–2018	NC	<LOD	<LOD	0.190 (<LOD–0.360)	0.390 (0.210–1.19)	407
6–11 years	2011–2012	0.241 (0.206–0.283)	0.220 (0.190–0.270)	0.450 (0.390–0.530)	0.930 (0.680–1.36)	1.37 (0.990–2.03)	401
	2013–2014	NC	<LOD	0.220 (0.150–0.310)	0.560 (0.340–0.840)	0.890 (0.640–1.10)	401
	2015–2016	NC	<LOD	<LOD	0.300 (0.200–0.380)	0.520 (0.360–0.700)	380
	2017–2018	NC	<LOD	<LOD	0.270 (0.210–0.550)	0.570 (0.430–0.879)	333
12–19 years	2011–2012	0.257 (0.212–0.312)	0.270 (0.220–0.340)	0.490 (0.390–0.600)	0.840 (0.650–1.24)	1.31 (0.920–1.75)	390
	2013–2014	NC	<LOD	0.240 (0.200–0.310)	0.560 (0.400–0.860)	1.02 (0.610–1.81)	452
	2015–2016	NC	<LOD	0.130 (<LOD–0.160)	0.350 (0.200–0.470)	0.610 (0.380–1.14)	402
	2017–2018	NC	<LOD	0.160 (<LOD–0.250)	0.330 (0.310–0.520)	0.700 (0.400–0.940)	364
≥20 years	2011–2012	0.346 (0.303–0.396)	0.340 (0.290–0.400)	0.720 (0.620–0.850)	1.49 (1.20–1.67)	1.93 (1.67–2.29)	1,716
	2013–2014	0.274 (0.246–0.305)	0.240 (0.200–0.280)	0.540 (0.450–0.630)	1.16 (1.00–1.33)	1.76 (1.44–2.04)	1,813
	2015–2016	NC	0.140 (0.130–0.150)	0.340 (0.310–0.380)	0.740 (0.650–0.890)	1.22 (0.970–1.43)	1,802
	2017–2018	NC	<LOD	0.370 (0.310–0.410)	0.750 (0.620–0.840)	1.16 (0.950–1.34)	1,708
Gender							
Males	2011–2012	0.342 (0.293–0.399)	0.330 (0.290–0.380)	0.670 (0.580–0.810)	1.34 (1.03–1.67)	1.91 (1.54–2.51)	1,260
	2013–2014	0.243 (0.219–0.268)	0.200 (0.170–0.220)	0.480 (0.390–0.600)	1.07 (0.840–1.33)	1.55 (1.28–1.96)	1,319
	2015–2016	NC	<LOD	0.280 (0.230–0.330)	0.620 (0.540–0.710)	0.900 (0.760–1.18)	1,533
	2017–2018	NC	<LOD	0.290 (0.220–0.380)	0.620 (0.550–0.780)	1.04 (0.760–1.39)	1,382
Females	2011–2012	0.307 (0.262–0.360)	0.300 (0.250–0.360)	0.660 (0.540–0.770)	1.37 (1.17–1.54)	1.82 (1.54–2.14)	1,247
	2013–2014	0.249 (0.218–0.284)	0.210 (0.170–0.260)	0.470 (0.390–0.570)	1.07 (0.820–1.27)	1.75 (1.25–2.26)	1,347
	2015–2016	NC	<LOD	0.290 (0.250–0.350)	0.730 (0.570–0.980)	1.27 (0.940–1.88)	1,547
	2017–2018	NC	<LOD	0.310 (0.230–0.390)	0.720 (0.590–0.820)	1.07 (0.840–1.25)	1,430

Table 5-31. Geometric Mean and Selected Percentiles of Total Mercury Urinary Concentrations (in µg/L) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Race/ethnicity							
Mexican Americans	2011–2012	0.301 (0.261–0.348)	0.300 (0.200–0.400)	0.620 (0.510–0.680)	1.25 (0.910–1.53)	1.75 (1.32–2.25)	317
	2013–2014	0.229 (0.198–0.265)	0.160 (0.150–0.210)	0.450 (0.300–0.620)	1.12 (0.780–1.35)	1.47 (0.970–2.38)	454
	2015–2016	NC	<LOD	0.230 (0.170–0.280)	0.530 (0.440–0.640)	0.900 (0.650–1.22)	586
	2017–2018	NC	<LOD	0.280 (0.200–0.400)	0.770 (0.480–1.36)	1.36 (0.720–3.95)	436
Non-Hispanic blacks	2011–2012	0.360 (0.316–0.410)	0.360 (0.320–0.400)	0.670 (0.570–0.800)	1.33 (1.06–1.60)	1.99 (1.48–3.06)	671
	2013–2014	0.279 (0.228–0.340)	0.250 (0.190–0.320)	0.530 (0.400–0.690)	1.10 (0.900–1.49)	1.82 (1.11–2.48)	580
	2015–2016	NC	0.140 (<LOD–0.180)	0.370 (0.270–0.510)	0.800 (0.620–1.19)	1.38 (0.870–1.72)	676
	2017–2018	NC	<LOD	0.370 (.0280–0.480)	0.900 (2.02–1.86)	1.31 (1.01–1.86)	641
Non-Hispanic whites	2011–2012	0.308 (0.260–0.365)	0.290 (0.260–0.360)	0.630 (0.510–0.810)	1.37 (1.09–1.64)	1.77 (1.49–2.14)	819
	2013–2014	0.240 (0.211–0.271)	0.200 (0.160–0.230)	0.460 (0.370–0.580)	1.06 (0.840–1.24)	1.64 (1.24–2.04)	988
	2015–2016	NC	<LOD	0.260 (0.210–0.330)	0.700 (0.480–0.870)	1.18 (0.860–1.40)	932
	2017–2018	NC	<LOD	0.280 (0.200–0.370)	0.620 (0.570–0.750)	0.840 (0.680–1.16)	918
All Hispanics	2011–2012	0.330 (0.299–0.364)	0.330 (0.270–0.390)	0.680 (0.610–0.760)	1.30 (1.15–1.53)	1.98 (1.61–2.42)	574
	2013–2014	0.239 (0.207–0.276)	0.180 (0.150–0.240)	0.460 (0.360–0.620)	1.14 (0.800–1.35)	1.57 (1.24–2.15)	702
	2015–2016	NC	<LOD	0.260 (0.210–0.330)	0.600 (0.470–0.740)	1.03 (0.730–1.37)	986
	2017–2018	NC	<LOD	0.310 (0.250–0.420)	0.810 (0.580–0.990)	1.34 (0.910–2.03)	677
Asians	2011–2012	0.430 (0.351–0.527)	0.450 (0.330–0.580)	0.910 (0.750–1.12)	1.69 (1.31–2.06)	2.41 (1.77–3.53)	355
	2013–2014	0.313 (0.269–0.363)	0.270 (0.220–0.340)	0.620 (0.520–0.710)	1.18 (0.890–1.66)	1.78 (1.20–3.10)	291
	2015–2016	NC	0.180 (<LOD–0.250)	0.390 (0.330–0.510)	0.720 (0.610–0.820)	0.960 (0.790–1.49)	333
	2017–2018	NC	0.190 (<LOD–0.280)	0.430 (0.350–0.600)	0.990 (0.640–1.29)	1.49 (1.05–2.46)	365

[a] The limits of detection for survey years 2011–2012, 2013–2014, 2015–2016, and 2017–2018 were 0.05, 0.13, 0.13 and 0.13 µg/L, respectively.

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

Source: CDC 2024

Table 5-32. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations (µg/g of Creatinine) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2003–2004	0.443 (0.404–0.486)	0.447 (0.392–0.498)	0.909 (0.785–1.00)	1.65 (1.40–1.86)	2.35 (1.88–2.85)	2,537
	2005–2006	0.460 (0.414–0.511)	0.450 (0.410–0.510)	0.870 (0.790–1.00)	1.63 (1.44–1.75)	2.26 (2.12–2.50)	2,578
	2007–2008	0.462 (0.425–0.502)	0.450 (0.400–0.490)	0.820 (0.750–0.960)	1.57 (1.38–1.73)	2.32 (2.00–2.89)	2,634
	2009–2010	NC	0.409 (0.367–0.459)	0.793 (0.691–0.893)	1.43 (1.24–1.67)	2.09 (1.79–2.39)	2,865
Age group							
6–11 years	2003–2004	0.297 (0.246–0.358)	0.276 (0.208–0.347)	0.485 (0.391–0.630)	1.25 (0.667–1.79)	1.79 (1.11–2.61)	286
	2005–2006	0.411 (0.323–0.524)	0.390 (0.290–0.500)	0.710 (0.510–0.960)	1.30 (0.990–2.12)	2.55 (1.38–3.50)	355
	2007–2008	0.393 (0.351–0.440)	0.350 (0.300–0.440)	0.630 (0.540–0.770)	1.15 (0.860–1.50)	1.68 (1.18–2.99)	398
	2009–2010	NC	0.357 (0.306–0.406)	0.632 (0.500–0.750)	1.04 (0.863–1.26)	1.62 (1.19–1.98)	379
12–19 years	2003–2004	0.255 (0.225–0.289)	0.217 (0.196–0.275)	0.464 (0.376–0.535)	1.06 (0.714–1.39)	1.67 (1.13–2.03)	722
	2005–2006	.286 (0.230–0.356)	0.260 (0.200–0.320)	0.500 (0.380–0.660)	1.09 (0.660–1.70)	1.76 (1.11–2.67)	703
	2007–2008	0.284 (0.251–0.320)	0.280 (0.230–0.300)	0.500 (0.400–0.550)	.890 (0.620–1.08)	1.18 (0.980–1.36)	375
	2009–2010	NC	0.226 (0.202–0.287)	0.481 (0.429–0.553)	.917 (0.736–1.18)	1.41 (1.12–1.62)	455
≥20 years	2003–2004	0.508 (0.455–0.566)	0.525 (0.447–0.616)	1.00 (0.875–1.09)	1.76 (1.46–2.11)	2.54 (2.04–3.00)	1,529
	2005–2006	0.503 (0.461–0.549)	0.510 (0.470–0.550)	0.940 (0.850–1.07)	1.69 (1.50–1.86)	2.31 (2.12–2.54)	1,520
	2007–2008	0.507 (0.463–0.555)	0.500 (0.450–0.550)	0.940 (0.810–1.02)	1.69 (1.51–2.01)	2.56 (2.09–3.17)	1,861
	2009–2010	NC	0.454 (0.395–0.517)	0.861 (0.731–0.988)	1.51 (1.29–1.85)	2.15 (1.88–2.57)	2,031
Gender							
Males	2003–2004	0.365 (0.333–0.400)	0.362 (0.309–0.417)	0.696 (0.620–0.784)	1.31 (1.18–1.44)	1.87 (1.51–2.30)	1,266
	2005–2006	0.380 (0.336–0.431)	0.390 (0.330–0.440)	0.740 (0.600–0.890)	1.27 (1.09–1.47)	1.73 (1.62–1.85)	1,270
	2007–2008	0.408 (0.374–0.445)	0.390 (0.350–0.450)	0.730 (0.650–0.810)	1.22 (1.11–1.36)	1.69 (1.54–2.11)	1,326
	2009–2010	NC	0.337 (0.298–0.391)	0.675 (0.585–0.802)	1.19 (1.06–1.29)	1.50 (1.33–1.78)	1,404
Females	2003–2004	0.532 (0.472–0.599)	0.545 (0.455–0.652)	1.06 (0.969–1.21)	1.88 (1.64–2.30)	2.77 (2.12–3.56)	1,271
	2005–2006	0.552 (0.494–0.617)	0.540 (0.490–0.620)	1.09 (0.850–1.27)	1.96 (1.72–2.14)	2.78 (2.35–3.17)	1,308
	2007–2008	0.520 (0.469–0.576)	0.490 (0.460–0.540)	0.960 (0.820–1.11)	1.92 (1.58–2.24)	2.83 (2.24–3.50)	1,308
	2009–2010	NC	0.475 (0.423–0.552)	0.890 (0.771–1.07)	1.81 (1.43–2.09)	2.57 (2.09–2.94)	1,461

Table 5-32. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations (µg/g of Creatinine) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2003–2010

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Race/ethnicity							
Mexican Americans	2003–2004	0.384 (0.307–0.480)	0.365 (0.280–0.455)	0.768 (0.619–0.990)	1.62 (1.23–2.16)	2.32 (1.78–4.01)	618
	2005–2006	0.425 (0.337–0.536)	0.400 (0.310–0.490)	0.840 (0.560–1.29)	1.82 (1.30–2.47)	2.63 (2.22–3.20)	651
	2007–2008	0.409 (0.350–0.479)	0.380 (0.310–0.480)	0.790 (0.690–0.850)	1.55 (1.08–1.98)	2.03 (1.55–2.70)	514
	2009–2010	NC	0.333 (0.272–0.400)	0.660 (0.494–0.861)	1.29 (1.02–1.54)	1.95 (1.52–2.89)	615
Non-Hispanic blacks	2003–2004	0.343 (0.301–0.391)	0.306 (0.265–0.368)	0.587 (0.522–0.687)	1.28 (0.964–1.63)	2.13 (1.41–2.87)	713
	2005–2006	0.328 (0.285–0.378)	0.320 (0.270–0.370)	0.610 (0.470–0.780)	1.15 (0.930–1.40)	1.64 (1.29–1.96)	691
	2007–2008	0.350 (0.303–0.404)	0.330 (0.280–0.380)	0.590 (0.490–0.690)	1.10 (0.840–1.46)	1.85 (1.13–2.77)	589
	2009–2010	NC	0.317 (0.259–0.393)	0.582 (0.500–0.659)	1.05 (0.900–1.30)	1.55 (1.18–1.96)	546
Non-Hispanic whites	2003–2004	0.463 (0.400–0.537)	0.476 (0.385–0.588)	0.970 (0.800–1.07)	1.67 (1.32–2.11)	2.40 (1.88–2.90)	1,066
	2005–2006	0.475 (0.426–0.531)	0.490 (0.440–0.540)	0.890 (0.820–1.02)	1.61 (1.42–1.75)	2.23 (1.98–2.50)	1,044
	2007–2008	0.481 (0.423–0.546)	0.480 (0.390–0.540)	0.890 (0.750–1.03)	1.58 (1.34–2.02)	2.49 (1.89–3.18)	1,100
	2009–2010	NC	0.434 (0.370–0.500)	0.833 (0.689–1.04)	1.50 (1.26–1.87)	2.12 (1.80–2.64)	1,225

[a] The limits of detection were based on non-creatinine corrected urinary concentrations, which were 0.14, 0.11, 0.08, and 0.08 µg/L for survey years 2003–2004, 2005–2006, 2007–2008, and 2009–2010, respectively.

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

Source: CDC 2024

Table 5-33. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations (µg/g of Creatinine) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Total	2011–2012	0.367 (0.333–0.405)	0.353 (0.306–0.394)	0.676 (0.623–0.754)	1.33 (1.13–1.50)	1.75 (1.49–2.32)	2,505
	2013–2014	0.283 (0.260–0.309)	0.270 (0.250–0.290)	0.571 (0.511–0.644)	1.20 (1.05–1.36)	1.61 (1.47–1.81)	2,665
	2015–2016	NC	<LOD	0.356 (0.318–0.391)	0.708 (0.628–0.817)	1.10 (0.912–1.25)	3,077
	2017–2018	NC	<LOD	0.362 (0.327–0.402)	0.683 (0.579–0.761)	0.962 (0.841–1.13)	2,810
Age group							
3–5 years	2015–2016	NC	<LOD	<LOD	0.667 (<LOD–0.818)	0.994 (0.818–1.13)	495
	2017–2018	NC	<LOD	<LOD	0.692 (<LOD–0.818)	0.923 (0.735–1.46)	407
6–11 years	2011–2012	0.345 (0.298–0.398)	0.306 (0.276–0.344)	0.537 (0.441–0.613)	1.08 (0.884–1.43)	1.62 (1.07–2.34)	400
	2013–2014	NC	<LOD	0.429 (0.310–0.529)	0.750 (0.563–0.897)	1.11 (0.713–1.72)	401
	2015–2016	NC	<LOD	<LOD	0.474 (0.409–0.529)	0.643 (0.500–1.00)	380
	2017–2018	NC	<LOD	<LOD	0.529 (0.450–0.745)	0.961 (0.635–1.29)	332
12–19 years	2011–2012	0.246 (0.219–0.277)	0.221 (0.190–0.269)	0.405 (0.368–0.453)	0.735 (0.571–1.11)	1.21 (0.742–1.49)	390
	2013–2014	NC	<LOD	0.257 (0.200–0.281)	0.580 (0.391–0.735)	0.846 (0.580–1.07)	452
	2015–2016	NC	<LOD	0.194 (<LOD–0.220)	0.320 (0.246–0.568)	0.650 (0.385–0.967)	402
	2017–2018	NC	<LOD	0.205 (<LOD–0.500)	0.353 (0.281–0.500)	0.642 (0.353–1.12)	364
≥20 years	2011–2012	0.393 (0.351–0.439)	0.383 (0.330–0.437)	0.750 (0.673–0.805)	1.38 (1.17–1.63)	1.95 (1.50–2.48)	1,715
	2013–2014	0.318 (0.291–0.349)	0.304 (0.281–0.333)	0.644 (0.561–0.741)	1.32 (1.13–1.47)	1.76 (1.50–1.88)	1,812
	2015–2016	NC	0.198 (0.184–0.210)	0.391 (0.349–0.449)	0.776 (0.692–0.886)	1.15 (1.00–1.32)	1,800
	2017–2018	NC	<LOD	0.388 (0.344–0.441)	0.714 (0.563–0.859)	1.00 (0.841–1.32)	1,707
Gender							
Males	2011–2012	0.320 (0.278–0.368)	0.294 (0.267–0.358)	0.558 (0.478–0.667)	1.11 (0.791–1.44)	1.57 (1.21–2.00)	1,259
	2013–2014	0.242 (0.223–0.263)	0.231 (0.206–0.259)	0.476 (0.429–0.542)	0.902 (0.779–1.11)	1.31 (1.13–1.49)	1,318
	2015–2016	NC	<LOD	0.274 (0.246–0.300)	0.563 (0.490–0.628)	0.818 (0.705–0.994)	1,533
	2017–2018	NC	<LOD	0.316 (0.265–0.360)	0.501 (0.433–0.623)	0.692 (0.551–0.948)	1,381
Females	2011–2012	0.418 (0.374–0.466)	0.409 (0.355–0.453)	0.800 (0.706–0.900)	1.46 (1.29–1.65)	2.00 (1.63–2.60)	1,246
	2013–2014	0.330 (0.297–0.367)	0.315 (0.273–0.356)	0.692 (0.600–0.822)	1.44 (1.18–1.68)	1.83 (1.60–2.12)	1,347
	2015–2016	NC	<LOD	0.415 (0.375–0.473)	0.875 (0.750–1.00)	1.30 (1.06–1.86)	1,544
	2017–2018	NC	<LOD	0.429 (0.368–0.482)	0.841 (0.701–0.948)	1.13 (1.00–1.45)	1,429

Table 5-33. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations (μg/g of Creatinine) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2018

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI)[a]				Sample size
			50th	75th	90th	95th	
Race/ethnicity							
Mexican Americans	2011–2012	0.339 (0.288–0.399)	0.286 (0.225–0.393)	0.641 (0.433–0.789)	1.17 (1.00–1.42)	1.70 (1.31–2.24)	317
	2013–2014	0.261 (0.231–0.295)	0.237 (0.209–0.273)	0.516 (0.409–0.709)	1.04 (0.810–1.48)	1.62 (1.11–2.55)	454
	2015–2016	NC	<LOD	0.283 (0.236–0.360)	0.692 (0.530–0.810)	1.04 (0.791–1.30)	585
	2017–2018	NC	<LOD	0.346 (0.257–0.447)	0.788 (0.477–1.22)	1.68 (0.745–4.66)	434
Non-Hispanic blacks	2011–2012	0.280 (0.245–0.320)	0.261 (0.224–0.294)	0.467 (0.411–0.529)	0.896 (0.638–1.14)	1.43 (1.10–1.57)	671
	2013–2014	0.211 (0.169–0.264)	0.202 (0.152–0.269)	0.409 (0.333–0.516)	0.794 (0.643–1.10)	1.34 (0.880–1.52)	580
	2015–2016	NC	0.136 (<LOD–0.163)	0.303 (0.237–0.367)	0.642 (0.450–0.848)	0.901 (0.695–1.18)	674
	2017–2018	NC	<LOD	0.294 (0.225–0.360)	0.616 (0.478–0.762)	0.889 (0.667–1.39)	641
Non-Hispanic whites	2011–2012	0.372 (0.323–0.428)	0.364 (0.294–0.433)	0.700 (0.619–0.805)	1.35 (1.05–1.63)	1.75 (1.41–2.48)	817
	2013–2014	0.295 (0.269–0.323)	0.278 (0.257–0.310)	0.602 (0.516–0.689)	1.27 (1.08–1.45)	1.64 (1.46–1.82)	987
	2015–2016	NC	<LOD	0.349 (0.300–0.400)	0.708 (0.563–0.886)	1.10 (0.827–1.32)	932
	2017–2018	NC	<LOD	0.368 (0.310–0.441)	0.621 (0.514–0.745)	0.867 (0.683–1.09)	918
All Hispanics	2011–2012	0.369 (0.342–0.399)	0.331 (0.283–0.384)	0.674 (0.612–0.772)	1.24 (1.13–1.44)	1.86 (1.47–2.92)	574
	2013–2014	0.267 (0.237–0.300)	0.237 (0.214–0.273)	0.541 (0.448–0.634)	1.07 (0.837–1.36)	1.61 (1.24–1.86)	702
	2015–2016	NC	<LOD	0.348 (0.284–0.408)	0.717 (0.574–0.917)	1.15 (0.948–1.36)	985
	2017–2018	NC	<LOD	0.358 (0.310–0.426)	0.788 (0.605–0.976)	1.17 (0.929–2.25)	675
Asians	2011–2012	0.577 (0.473–0.705)	0.562 (0.467–0.700)	1.16 (0.872–1.44)	1.82 (1.54–2.00)	2.29 (1.90–3.12)	355
	2013–2014	0.488 (0.422–0.565)	0.475 (0.373–0.600)	0.917 (0.779–1.06)	1.88 (1.35–2.19)	2.57 (1.88–4.24)	291
	2015–2016	NC	0.290 (<LOD–0.375)	0.594 (0.457–0.706)	1.02 (0.808–1.22)	1.38 (1.11–1.80)	333
	2017–2018	NC	0.300 (<LOD–0.367)	0.566 (0.474–0.731)	1.05 (0.848–1.45)	1.47 (1.07–1.94)	365

[a] The limits of detection were based on non-creatinine corrected urinary concentrations, which were 0.05, 0.13, 0.13, and 0.13 μg/L for survey years 2011–2012, 2013–2014, 2015–2016, and 2017–2018 respectively.

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

Source: CDC 2024

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

In addition to workers exposed to mercury (primarily mercury vapor) in the workplace, other population groups are at risk for potential exposure to high levels of mercury. These include: (1) people who live in proximity to former mercury mining or production sites, secondary production (recycling) facilities, municipal or medical incinerators, or coal-fired power plants; (2) people who consume large amounts of fish or marine mammals (Grandjean et al. 1992); (3) people who have mercury amalgam dental restorations (Mackert and Berglund 1997; Snapp et al. 1989); (4) people who use consumer products containing mercury such as traditional or herbal remedies, or cosmetics, including skin lightening creams (McKelvey et al. 2011); and (5) people who use mercury as part of spiritual practices (Riley et al. 2001).

Mercury levels were measured in air samples at several recycling operations in Brazil (Gouveia et al. 2019). Highest levels tended to be near the recycling piles (0.032 $\mu g/m^3$) and conveyer belts (0.032 $\mu g/m^3$); however, only 14.5% of the samples had levels above the limit of quantification. People involved in artisanal and small-scale gold mining are potentially exposed to higher levels of mercury than the general population. Total mercury and methylmercury levels in blood, urine, and hair, were monitored from 238 participants with occupational exposure to mercury from ASGM activities in Colombia (Calao-Ramos et al. 2021). The median values of total mercury in blood (3.70 μg/L), urine (4.00 μg/L), and hair (1.37 mg/kg), and methylmercury concentrations in hair (1.47 mg/kg) for the study group were below allowed concentrations set by WHO. The study results did, however, show that 40% of the miners had levels in blood, urine, and/or hair above the WHO thresholds. Burning of amalgam and fish consumption was correlated with the highest mercury levels in biological matrices.