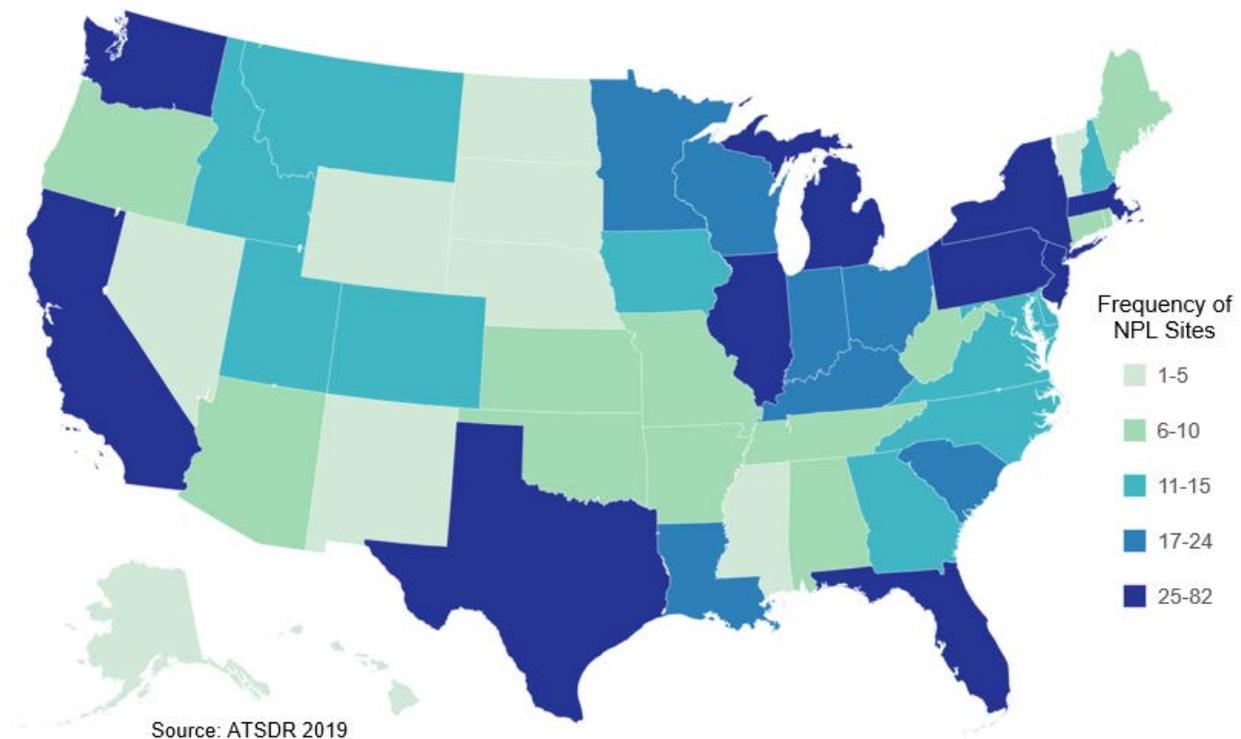


## CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

### 5.1 OVERVIEW

Mercury or mercury compounds have been identified in at least 838 of the 1,867 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2019). 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, 828 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**



- The general population is primarily exposed to mercury through the ingestion of foods.
- 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.

## 5. POTENTIAL FOR HUMAN EXPOSURE

- 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.

## 5. POTENTIAL FOR HUMAN EXPOSURE

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 the earth's crust at levels averaging 0.5 ppm, but the actual concentration varies considerably depending on location (Budavari 1989; Sidle 1993). 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

## 5. POTENTIAL FOR HUMAN EXPOSURE

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 2020). 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). The USGS reported that, in 2019, 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. 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). China is the overwhelming producer of mined mercury (~3,500 metric tons in 2019). Other nations with mine production of mercury include: Argentina, Kyrgyzstan, Mexico, Norway, Peru, and Tajikistan (USGS 2020). Artisanal goldmining is practiced in many countries. In artisanal mining, gold is extracted from a substrate (e.g., pulverized ore, sediment, soil) by mixing the substrate with elemental mercury to form mercury-gold amalgam. The amalgam is washed, sedimented, and roasted to vaporize the elemental mercury out of the amalgam. This process can result in direct exposures of mine workers, gold shop merchants, and nearby households to mercury vapor and can release mercury to the environment where it can be converted to methylmercury (Counter et al. 1998; Diringer et al. 2015; Moody et al. 2020; Ramirez et al. 2000; Salazar-Camacho et al. 2021)

Tables 5-1 and 5-2 list the facilities in each state that process or recycle mercury and mercury compounds, respectively. Data from the TRI is intended to meet the requirements of the Emergency Planning and Community Right-to-Know Act (EPCRA) Section 313 for mercury and the mercury compounds category.

## 5. POTENTIAL FOR HUMAN EXPOSURE

If a facility processes or imports mercury or mercury containing compounds, it is required to report these data to the TRI and this meets the definition of the reporting category, Produce.

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

State <sup>a</sup>	Number of facilities	Minimum amount on site in pounds <sup>b</sup>	Maximum amount on site in pounds <sup>b</sup>	Activities and uses <sup>c</sup>
AK	1	0	99	1, 5, 12
AL	7	0	999	1, 5, 12, 13, 14
AR	4	0	99	1, 5, 12, 13, 14
AZ	8	0	9,999	1, 4, 5, 6, 8, 12, 13, 14
CA	31	0	99,999	1, 2, 5, 7, 8, 9, 10, 11, 12, 13, 14
CO	6	0	99	1, 5, 8, 12, 13, 14
CT	3	0	999	8, 14
DC	1	0	99	7
DE	1	1,000	9,999	8, 9
FL	6	0	9,999	1, 4, 5, 8, 9, 12, 13, 14
GA	6	0	999	1, 2, 8, 12, 13, 14
HI	1	0	99	1, 5
IA	5	0	9,999	1, 5, 7, 8, 12, 13, 14
ID	3	0	999	1, 5, 12, 13, 14
IL	19	0	9,999	1, 5, 6, 8, 12, 13, 14
IN	8	0	9,999	1, 2, 5, 6, 8, 12, 13, 14
KS	4	0	999	1, 2, 5, 8, 12, 13, 14
KY	9	0	99,999	1, 5, 8, 10, 12, 13, 14
LA	14	0	9,999,999	1, 2, 3, 5, 12, 13, 14
MA	7	0	9,999	1, 5, 8, 11, 12, 13, 14
MD	2	0	99	14
ME	2	0	99	12, 14
MI	5	0	9,999	1, 5, 8, 9, 13, 14
MN	6	0	9,999	1, 5, 8, 12, 13, 14
MO	2	0	99	1, 5, 14
MP	1	0	99	1, 13
MS	3	0	99	1, 5, 9, 12, 14
MT	2	0	99	1, 10, 13, 14
NC	4	0	999	8, 12, 14
NE	9	0	999	1, 3, 4, 5, 9, 12, 13, 14
NJ	1	0	99	12
NM	1	10,000	99,999	1, 5, 12
NV	9	0	9,999,999	1, 8, 9, 12, 13, 14
NY	6	0	9,999	1, 11, 12, 13, 14

## 5. POTENTIAL FOR HUMAN EXPOSURE

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

State <sup>a</sup>	Number of facilities	Minimum amount on site in pounds <sup>b</sup>	Maximum amount on site in pounds <sup>b</sup>	Activities and uses <sup>c</sup>
OH	20	0	9,999,999	1, 2, 3, 5, 7, 8, 9, 10, 12, 13, 14
OK	4	0	99	1, 2, 5, 8, 9, 12, 13, 14
OR	2	0	999	12, 14
PA	17	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
RI	1	0	99	12, 14
SC	3	0	99	1, 5, 8, 12, 14
SD	1	0	99	1, 5, 14
TN	7	0	9,999,999	1, 2, 3, 5, 9, 12, 13, 14
TX	44	0	99,999	1, 2, 4, 5, 7, 8, 9, 11, 12, 13, 14
UT	5	0	99	1, 5, 12, 14
VA	4	0	9,999	1, 5, 7, 12, 14
VT	1	0	99	8
WA	1	100	999	14
WI	8	0	999,999	1, 3, 5, 7, 9, 12, 14
WV	10	0	9,999	1, 5, 12, 13, 14
WY	5	0	9,999	1, 2, 5, 8, 9, 12, 13, 14

<sup>a</sup>Post office state abbreviations used.

<sup>b</sup>Amounts on site reported by facilities in each state.

<sup>c</sup>Activities/Uses:

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

Source: TRI19 2020 (Data are from 2019)

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

State <sup>a</sup>	Number of facilities	Minimum amount on site in pounds <sup>b</sup>	Maximum amount on site in pounds <sup>b</sup>	Activities and uses <sup>c</sup>
AK	14	0	999,999	1, 2, 5, 7, 9, 12, 13, 14
AL	34	0	99,999	1, 3, 4, 5, 6, 8, 12, 13, 14
AR	18	0	99,999	1, 2, 3, 4, 5, 6, 7, 9, 10, 11, 12, 13, 14
AZ	21	0	99,999	1, 2, 3, 4, 5, 8, 9, 12, 13, 14
CA	70	0	99,999	1, 2, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
CO	19	0	9,999,999	1, 4, 5, 6, 7, 8, 9, 12, 13, 14
CT	2	0	999	1, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
DC	1	0	99	14

## 5. POTENTIAL FOR HUMAN EXPOSURE

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

State <sup>a</sup>	Number of facilities	Minimum amount on site in pounds <sup>b</sup>	Maximum amount on site in pounds <sup>b</sup>	Activities and uses <sup>c</sup>
DE	3	0	99	1, 2, 3, 5, 7, 12, 13, 14
FL	33	0	99,999	1, 2, 3, 4, 5, 8, 9, 11, 12, 13, 14
GA	21	0	999	1, 2, 3, 4, 5, 8, 12, 13, 14
GU	1	0	99	7, 9
HI	7	0	99	1, 2, 5, 9, 12, 13, 14
IA	26	0	9,999	1, 3, 4, 5, 7, 9, 12, 13, 14
ID	11	0	9,999	1, 5, 7, 8, 11, 12, 13, 14
IL	38	0	9,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
IN	38	0	9,999	1, 2, 3, 4, 5, 7, 9, 11, 12, 13, 14
KS	21	0	9,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
KY	23	0	9,999	1, 2, 3, 4, 5, 8, 9, 10, 12, 13, 14
LA	46	0	99,999	1, 2, 3, 4, 5, 9, 10, 11, 12, 13, 14
MA	1	100	999	9
MD	13	0	999	1, 3, 4, 5, 9, 12, 13, 14
ME	3	0	999	1, 5, 12, 14
MI	30	0	99,999	1, 4, 5, 7, 8, 9, 12, 13, 14
MN	16	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
MO	29	0	9,999	1, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
MP	1	0	99	7, 9
MS	11	0	999	1, 2, 3, 4, 5, 8, 12, 13, 14
MT	15	0	99	1, 2, 3, 4, 5, 6, 9, 12, 13, 14
NC	33	0	999,999	1, 2, 3, 5, 7, 8, 9, 12, 13, 14
ND	12	0	999,999	1, 4, 5, 9, 12, 13, 14
NE	10	0	9,999	1, 2, 3, 5, 7, 9, 12, 13, 14
NH	2	0	99	1, 2, 3, 5, 9, 12, 13, 14
NJ	15	0	9,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
NM	9	0	999	1, 3, 4, 5, 9, 12, 13, 14
NV	39	0	49,999,999	1, 2, 3, 5, 9, 12, 13, 14
NY	13	0	9,999	1, 5, 6, 8, 12, 13, 14
OH	39	0	9,999	1, 2, 3, 4, 5, 7, 9, 11, 12, 13, 14
OK	24	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
OR	9	0	99	1, 5, 8, 12, 13, 14
PA	51	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
PR	5	0	99	1, 3, 5, 11, 12, 13, 14
RI	2	0	999	7, 14
SC	25	0	999,999	1, 3, 4, 5, 6, 7, 8, 10, 12, 13, 14
SD	3	0	99,999	1, 5, 9, 12, 13, 14
TN	26	0	9,999	1, 2, 5, 8, 9, 10, 11, 12, 13, 14

## 5. POTENTIAL FOR HUMAN EXPOSURE

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

State <sup>a</sup>	Number of facilities	Minimum amount on site in pounds <sup>b</sup>	Maximum amount on site in pounds <sup>b</sup>	Activities and uses <sup>c</sup>
TX	83	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
UT	23	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
VA	17	0	999	1, 3, 4, 5, 6, 7, 9, 10, 12, 13, 14
WA	23	0	999	1, 2, 3, 5, 7, 8, 9, 10, 12, 13, 14
WI	28	0	9,999	1, 3, 4, 5, 7, 8, 9, 12, 13, 14
WV	19	0	999,999	1, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
WY	13	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14

<sup>a</sup>Post office state abbreviations used.

<sup>b</sup>Amounts on site reported by facilities in each state.

<sup>c</sup>Activities/Uses:

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

Source: TRI19 2020 (Data are from 2019)

### 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).

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). 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

## 5. POTENTIAL FOR HUMAN EXPOSURE

2020b). 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 most important domestic end users of mercury in 2019 were those involved in the production of chlorine-caustic soda (chloralkali), dental products, electronics, and fluorescent-lighting manufacturing industries (USGS 2020). In 2020, the use of mercury in the production of chloralkali was decreased when one of the two operating facilities in the United States converted to a non-mercury process. 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, 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 20<sup>th</sup> century (Beauchamp et al. 2021; Byrns and Penning 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” (see Section 2.16).

***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).

## 5. POTENTIAL FOR HUMAN EXPOSURE

**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). 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).

**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.

## 5. POTENTIAL FOR HUMAN EXPOSURE

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

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  $3,100 \pm 1,900$  kg/year (3.1 metric tons/year) (Webster et al. 2016).

## 5. POTENTIAL FOR HUMAN EXPOSURE

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
<b>Total Gg (metric tons)</b>	<b>472 (472,000)</b>	<b>1,070 (1,070,000)</b>	<b>1,540 (1,540,000)</b>

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

## 5. POTENTIAL FOR HUMAN EXPOSURE

## 5.3.1 Air

Estimated releases of 4,409 pounds (~2 metric tons) of elemental mercury to the atmosphere from 329 facilities reporting to TRI domestic manufacturing and processing facilities in 2019, accounted for about 3.3% of the estimated total environmental releases from facilities required to report to the TRI (TRI19 2020). Another 31,119 pounds (~14 metric tons) of mercury compounds released to the atmosphere from 1,063 facilities reporting to TRI domestic manufacturing and processing facilities in 2019, accounted for about 1% of the estimated total environmental releases from facilities required to report to the TRI (TRI19 2020). 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<sup>a</sup>**

State <sup>c</sup>	RF <sup>d</sup>	Reported amounts released in pounds (metric tons) per year <sup>b</sup>								
		Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	Total release			
							On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site	
AK	1	10	0	0	28	0	10	28	38	
AL	7	653	0	0	7	0	657	3	660	
AR	4	45	0	0	191	0	236	0	236	
AZ	8	33	0	0	64	111	76	132	208	
CA	31	335	8	0	30,026	10	29,605	774	30,379	
CO	6	51	0	0	14	1	64	1	65	
CT	3	1	0	0	0	7	1	7	8	
DC	1	0	0	0	0	0	0	0	0	
DE	1	1	0	0	0	0	1	0	1	
FL	6	76	0	0	125	0	176	25	201	
GA	6	3	0	0	16	110	3	126	129	
HI	1	14	0	0	0	0	14	0	14	
IA	5	293	0	7	0	12,458	300	12,458	12,758	
ID	3	1	0	0	230	0	231	0	231	
IL	19	16	2	0	3,114	0	3,089	43	3,132	
IN	8	50	4	0	186	7	224	23	247	
KS	4	37	0	0	522	96	38	106	144	
KY	9	12	1	0	6,139	268	916	293	1,208	
LA	14	236	187	0	1,791	14	7,896	56	7,952	
MA	7	0	0	0	9	6	0	8	8	
MD	2	1	0	0	1	0	1	1	2	

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-4. Releases to the Environment from Facilities that Produce, Process, or Use Elemental Mercury<sup>a</sup>**

State <sup>c</sup>	RF <sup>d</sup>	Reported amounts released in pounds (metric tons) per year <sup>b</sup>								
		Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	Total release			
							On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site	
ME	2	0	0	0	0	0	0	0	0	
MI	5	72	0	0	1	270	79	270	349	
MN	6	6	0	0	35	0	7	35	42	
MO	2	13	0	0	25	0	32	6	39	
MP	1	45	0	14	12,217	0	45	0	45	
MS	3	5	0	0	2	0	5	2	7	
MT	2	1	0	0	116	0	1	0	1	
NC	4	0	0	0	2	0	0	1	1	
NE	8	91	0	0	2,336	108	208	132	339	
NJ	1	3	0	0	0	0	3	0	3	
NM	1	1	0	0	1,401	12	1	112	113	
NV	9	28	0	0	104	0	2,353	91	2,444	
NY	6	14	7	0	2	26	1,314	35	1,349	
OH	20	1,337	13	0	1,362	187	4,535	468	5,003	
OK	4	11	0	0	416	0	13	8	21	
OR	2	3	0	0	0	418	418	418	836	
PA	17	97	589	0	99	35,307	1,091	35,405	36,497	
RI	1	0	0	0	1	0	0	1	1	
SC	3	27	0	0	5	0	34	1	35	
SD	1	30	0	0	80	0	31	0	31	
TN	7	32	10	0	8,712	62	42	8,774	8,817	
TX	44	231	6	0	90	54	12,503	109	12,612	
UT	5	95	0	0	6	1	175	7	182	
VA	4	3	0	0	0	0	3	0	3	
VT	1	0	0	0	2,411	0	0	0	0	
WA	1	0	0	0	398	0	0	0	0	
WI	8	21	0	0	2,804	1,973	21	2,259	2,280	
WV	10	18	0	0	956	0	3,057	328	3,385	

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-4. Releases to the Environment from Facilities that Produce, Process, or Use Elemental Mercury<sup>a</sup>**

Reported amounts released in pounds (metric tons) per year <sup>b</sup>									
State <sup>c</sup>	RF <sup>d</sup>	Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	Total release		
							On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site
WY	5	357	0	1	140	0	887	9	896
Total	329	4,409	831	22	76,181	51,507	70,395	62,555	132,950
		(2)	(0.4)	(0.01)	(35)	(23)	(32)	(28)	(60)

<sup>a</sup>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.

<sup>b</sup>Data in TRI are maximum amounts released by each facility.

<sup>c</sup>Post office state abbreviations are used.

<sup>d</sup>Number of reporting facilities.

<sup>e</sup>The sum of fugitive and point source releases are included in releases to air by a given facility.

<sup>f</sup>Surface water discharges, wastewater treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).

<sup>g</sup>Class I wells, Class II-V wells, and underground injection.

<sup>h</sup>Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other onsite landfills, land treatment, surface impoundments, other land disposal, other landfills.

<sup>i</sup>Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown.

<sup>j</sup>The sum of all releases of the chemical to air, land, water, and underground injection wells.

<sup>k</sup>Total amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI19 2020 (Data are from 2019)

**Table 5-5. Releases to the Environment from Facilities that Produce, Process, or Use Mercury Compounds<sup>a</sup>**

Reported amounts released in pounds (metric tons) per year <sup>b</sup>									
State <sup>c</sup>	RF <sup>d</sup>	Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	Total release		
							On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site
AK	14	80	0	0	431,368	202	431,438	213	431,650
AL	34	1,383	13	0	14,861	62	15,413	904	16,318
AR	17	1,589	10	0	84,508	1	86,061	47	86,108
AZ	20	521	1	0	42,329	2	42,834	18	42,852
CA	65	665	44	0	20,920	230	21,416	443	21,859
CO	19	440	3	0	3,350	34	3,645	182	3,827
CT	2	0	0	0	0	0	0	0	0
DC	1	4	0	0	0	0	4	0	4
DE	3	105	1	0	22	1	128	1	129
FL	33	729	976	1	13,999	17	15,567	154	15,722
GA	21	295	27	0	339	2	553	110	663

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**Table 5-5. Releases to the Environment from Facilities that Produce, Process, or Use Mercury Compounds<sup>a</sup>**

State <sup>c</sup>	RF <sup>d</sup>	Reported amounts released in pounds (metric tons) per year <sup>b</sup>							
		Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	Total release		
							On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site
GU	1	0	0	0	0	0	0	0	0
HI	7	31	2	5	17	1	38	18	56
IA	26	226	8	0	1,298	75	935	672	1,607
ID	11	1,023	2	0	2,917	0	3,941	1	3,942
IL	36	1,045	34	0	3,014	6,213	3,645	6,661	10,306
IN	37	2,050	14	0	2,683	200	4,496	451	4,947
KS	21	203	7	0	559	19,679	677	19,771	20,448
KY	23	1,069	8	0	2,783	182	3,768	273	4,041
LA	43	394	13	4	1,468	19	1,846	52	1,898
MA	1	0	0	0	0	0	0	0	0
MD	13	137	5	0	517	28	206	480	686
ME	3	32	0	0	22	0	43	11	54
MI	29	338	13	0	38,686	17	7,094	31,960	39,054
MN	16	196	1	0	890	5	915	177	1,092
MO	28	1,071	6	1	1,106	2,094	2,157	2,121	4,278
MP	1	0	0	0	0	0	0	0	0
MS	11	412	199	22	428	42	701	402	1,104
MT	14	192	1	18	1,091	14	1,268	48	1,316
NC	33	1,200	9	0	36,820	44	3,014	35,058	38,072
ND	11	1,017	6	0	1,269	34	2,180	146	2,326
NE	10	240	2	0	24,378	97	18,707	6,011	24,718
NH	2	3	0	0	110	0	3	110	113
NJ	15	29	52	0	405	707	33	1,160	1,194
NM	9	58	0	2	846	630	906	630	1,536
NV	34	2,935	0	0	4,019,290	330,653	4,007,357	345,523	4,352,879
NY	13	96	0	0	1,723	30	1,818	32	1,850
OH	39	987	17	120	2,759	247	3,530	601	4,130
OK	23	350	4	0	2,095	378	2,240	587	2,827
OR	9	64	4	0	1,958	1	222	1,804	2,027
PA	51	1,676	12	0	4,211	178	3,768	2,310	6,078
PR	5	122	2	0	16	0	125	16	141
RI	2	1	0	0	0	0	1	0	1
SC	25	1,351	12	0	204	115	1,525	157	1,682
SD	3	22	0	0	7,045	0	7,067	0	7,067
TN	26	749	451	0	1,585	36	2,775	46	2,821
TX	83	3,521	235	99	69,103	6,569	70,343	9,183	79,526

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**Table 5-5. Releases to the Environment from Facilities that Produce, Process, or Use Mercury Compounds<sup>a</sup>**

State <sup>c</sup>	RF <sup>d</sup>	Reported amounts released in pounds (metric tons) per year <sup>b</sup>							
		Air <sup>e</sup>	Water <sup>f</sup>	UI <sup>g</sup>	Land <sup>h</sup>	Other <sup>i</sup>	Total release		
							On-site <sup>j</sup>	Off-site <sup>k</sup>	On- and off-site
UT	21	675	382	0	87,268	1	80,213	8,114	88,327
VA	17	380	5	0	350	8	713	30	743
WA	23	315	3	0	324	133	583	191	774
WI	27	238	3	0	1,629	2,316	901	3,285	4,186
WV	19	503	17	0	5,296	76	4,694	1,197	5,891
WY	13	356	0	0	767	11	881	254	1,134
Total	1,063	31,119 (14)	2,602 (1)	271 (0.1)	4,938,627 (2,240)	371,384 (168)	4,862,388 (2,206)	481,615 (218)	5,344,003 (2,424)

<sup>a</sup>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.

<sup>b</sup>Data in TRI are maximum amounts released by each facility.

<sup>c</sup>Post office state abbreviations are used.

<sup>d</sup>Number of reporting facilities.

<sup>e</sup>The sum of fugitive and point source releases are included in releases to air by a given facility.

<sup>f</sup>Surface water discharges, wastewater treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).

<sup>g</sup>Class I wells, Class II-V wells, and underground injection.

<sup>h</sup>Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other onsite landfills, land treatment, surface impoundments, other land disposal, other landfills.

<sup>i</sup>Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown.

<sup>j</sup>The sum of all releases of the chemical to air, land, water, and underground injection wells.

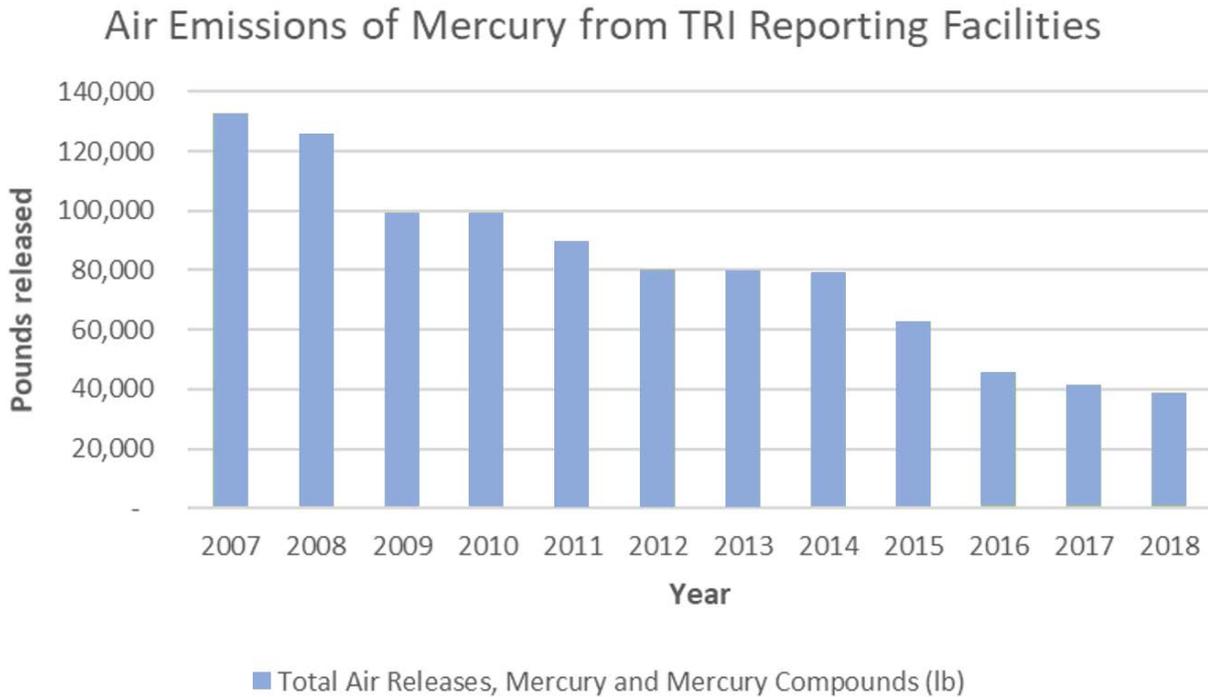
<sup>k</sup>Total amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI19 2020 (Data are from 2019)

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).

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**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). 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).

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**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

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**5.3.2 Water**

Estimated releases of 378 pounds (~0.4 metric tons) of elemental mercury to water from 329 facilities reporting to TRI domestic manufacturing and processing facilities in 2019, accounted for <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI19 2020).

Another 2,691 pounds (~1.2 metric tons) of mercury compounds that were released to the water from 1,063 facilities reporting to TRI domestic manufacturing and processing facilities in 2019, accounted for <1% of the estimated total environmental releases from facilities required to report to the TRI (TRI19 2020). 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%).

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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 ( $\mu\text{g}/\text{m}^2$ )
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 76,181 pounds (~34.6 metric tons) of elemental mercury to soil from 329 facilities reporting to TRI domestic manufacturing and processing facilities in 2019, accounted for about 57% of the estimated total environmental releases from facilities required to report to the TRI (TRI19 2020). Another 4,938,627 pounds (~2,240.8 metric tons) of mercury compounds released to soil from 1,063 facilities reporting to TRI domestic manufacturing and processing facilities in 2019, accounted for over 90% of the estimated total environmental releases from facilities required to report to the TRI (TRI19 2020). These releases are summarized in Tables 5-4 and 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 (MDN 2020; Sato and Sada 1992; WHO 1990, 1991). Since vapor-phase mercury is subject to long-range transport, it can be deposited to remote areas such as

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the Arctic. Atmospheric deposition of mercury in the Arctic was estimated as  $133\pm 9$  metric tons per year in 1990 and decreased to  $119\pm 10$  metric tons in 2005 (Goodsite et al. 2013). 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  $11.7 \mu\text{g}/\text{m}^2\text{-year}$  (range  $2.2\text{--}23.4 \mu\text{g}/\text{m}^2\text{-year}$ ). 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.

Additional anthropogenic releases of mercury to soil are expected as a result of the disposal of industrial and domestic solid waste products (e.g., thermometers, electrical switches, and batteries) to landfills (see Table 5-8). Another source of mercury releases to soil is the disposal of municipal incinerator ash in landfills (Mumma et al. 1990). In 1987, nationwide concentrations of mercury present in the ash from municipal waste incineration ranged from 0.03 to 25 ppm (Mumma et al. 1990). Such releases may exhibit a seasonal variability. For example, fly ash collected prior to Christmas in December of 1989 contained significantly less mercury (6.5 ppm) than ash collected after Christmas in January of 1990 (45–58 ppm), possibly as a result of the increased use and disposal of batteries containing mercury in toys and other equipment used during this season (Mumma et al. 1991). Emission sources include stack emissions, ashes collected at the stack, ashes from electrostatic precipitators, and slags (Morselli et al. 1992). An analysis of mercury concentrations in soil, refuse combustibles, and bottom and fly ash from incinerators showed increasing concentrations of 0, 2, 4, and 100 mg/kg (ppm), respectively (Goldin et al. 1992).

**Table 5-8. Estimated Discards of Mercury in Products in Municipal Solid Waste<sup>a</sup>**

Products	Amount in tons (metric tons) <sup>b</sup>						
	1970	1975	1980	1985	1989	1995	2000
<b>Batteries</b>							
Alkaline	4.1	38.4	158.2	352.3	419.4	41.6	0.0
Mercuric oxide	301.9	287.8	266.8	235.2	196.6	131.5	98.5 <sup>c</sup>
Others	4.8	4.7	4.5	4.5	5.2	3.5	0.0
Subtotal batteries	310.8	330.9	429.5	592.0	621.2	176.6	98.5
<b>Electric lighting</b>							
Fluorescent lamps	18.9	21.5	1.1	0.7	0.8	1.0	11.6 <sup>d</sup>
High intensity lamps	0.2	0.3	23.2	27.9	26.0	14.7	1.2
Subtotal lighting	19.1	21.8	24.3	28.6	26.7	15.7	12.6

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**Table 5-8. Estimated Discards of Mercury in Products in Municipal Solid Waste<sup>a</sup>**

Paint residues	30.2	37.3	26.7	31.4	18.2	2.3	0.5
Fever thermometers	12.2	23.2	25.7	32.5	16.3	16.9	16.8
Thermostats	5.3	6.8	7.0	9.5	11.2	8.1	10.3
Pigments	32.3	27.5	23.0	25.2	10.0	3.0	1.5
Dental uses	9.3	9.7	7.1	6.2	4.0	2.9	2.3
Special paper coating	0.1	0.6	1.2	1.8	1.0	0.0	0.0
Mercury light switches	0.4	0.4	0.4	0.4	0.4	1.9	1.9
Film pack batteries	2.1	2.3	2.6	2.8	0.0	0.0	0.0
Subtotal other sources	91.8	107.8	83.7	109.8	61.1	35.1	33.3
Total discards	421.7 (382.6)	460.5 (417.8)	537.5 (487.6)	730.4 (662.6)	709.0 (643.2)	227.4 (206.3)	144.4 (131.0)

<sup>a</sup>EPA (1992a) (except fluorescent lamps estimates).

<sup>b</sup>Discards before recovery; 1 ton equals 2,000 pounds.

<sup>c</sup>The estimates for the years 1995–2000 do not reflect state, federal, or battery manufacturers' efforts to reduce the mercury content of batteries. Since 1992, several states have restricted mercury use in batteries and/or banned the sale of mercuric oxide batteries. Federal legislation to restrict mercury use in batteries is pending. The battery industry has eliminated mercury as an intentional additive in alkaline batteries, except in button cells.

<sup>d</sup>The estimated contribution of mercury from fluorescent lamps disposal to municipal solid waste was calculated based on industry estimates of a 4% growth rate in sales in conjunction with a 53% decrease in mercury content between 1989 and 1995, and a further 34% decrease in mercury content by the year 2000 (to 15 mg mercury per 4-foot fluorescent lamp) (National Electric Manufacturers Association 1995).

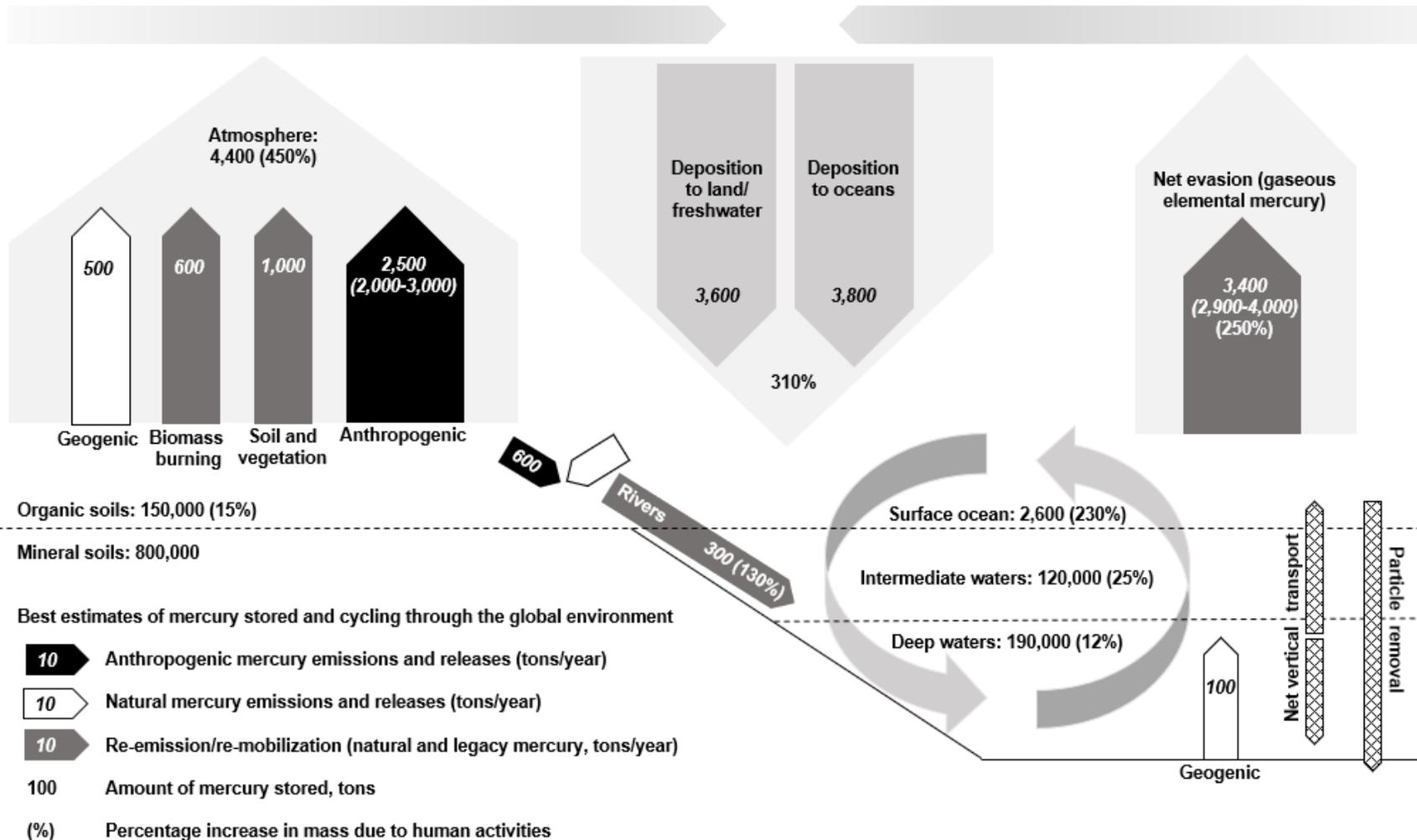
Source: EPA 1996

## 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 (UNEP 2018). 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).

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**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)

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**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. Over 95% of the mercury found in the atmosphere is gaseous mercury ( $\text{Hg}^0$ ), 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). Over the last 140 years, the atmospheric mercury concentrations have increased by a factor of 3.7, or approximately 2% per year (Swain et al. 1992).

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 ( $\text{Hg}^{+2}$ ) in precipitation is bound to aerosol particulates, which are relatively immobile when deposited on soil or water (Meili et al. 1991). 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).

**Water.** In soils and surface waters, mercury can exist in the mercuric ( $\text{Hg}^{+2}$ ) and mercurous ( $\text{Hg}^{+1}$ ) states as a number of complex ions with varying water solubilities. Mercuric mercury, present as complexes

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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). 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  $\text{Hg}^{+2}$  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  $10.3 \mu\text{g}/\text{m}^2/\text{year}$  of mercury deposited) is absorbed to the soil and does not leach down into the groundwater (Krabbenhoft and Babiartz 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 Brummer 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 in soil is decreased with increasing pH and/or chloride ion concentrations (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

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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<sup>3</sup> 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 reabsorbed 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 5-9. The bioaccumulation potential for methylmercury in fish is influenced by

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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 5-9. Bioconcentration of Various Mercury Compounds by Freshwater and Saltwater Organisms**

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

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**Table 5-9. Bioconcentration of Various Mercury Compounds by Freshwater and Saltwater Organisms**

Organomercury compounds				
Eastern oyster (adult) <i>C. virginica</i>	Soft parts	Methylmercuric chloride	74	40,000
Eastern oyster (adult) <i>C. virginica</i>	Soft parts	Phenylmercuric chloride	74	40,000

<sup>a</sup>Results are based on the concentration of mercury, not the concentration of the mercury compound to which the animal was exposed.

<sup>b</sup>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 5-10.

**Table 5-10. Mercury Concentrations Found in Fish**

Species	Total mercury median (±SD) µg/kg	Trophic level
Doncella <i>Ageneiosus pardalis</i>	678.5±344.9	3.8
Moncholo <i>Hoplías malabaricus</i>	401.4±278.5	4.5
Caga <i>Trachelyopterus fisheri</i>	374.3±250.0	3.5
Bagre sapo <i>Pseudopimelodus schultzi</i>	432.7±897.0	3.7
Agujeta <i>Ctenolucius beani</i>	270.9±289.1	4
Majarra Negra <i>Caquetaia umbrifera</i>	218.5±221.1	3.8
Mayupa <i>Sternopygus macrurus</i>	177.0±452.2	3.2
Mojarra Amarilla <i>Caquetaia kraussii</i>	218.0±200.6	3.4
Liso <i>Rhamdia quelen</i>	145.8±211.6	3.9

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**Table 5-10. Mercury Concentrations Found in Fish**

Species	Total mercury median ( $\pm$ SD) $\mu$ g/kg	Trophic level
Sardina colirroja <i>Astyanax fasciatus</i>	117.2 $\pm$ 113.7	3
Charre <i>Pimelodus punctatus</i>	100.5 $\pm$ 66.9	3.3
Cocobolo <i>Andinoacara pulcher</i>	116.7 $\pm$ 59.7	3.3
Dentón <i>Leporinus muyscorum</i>	116.7 $\pm$ 68.0	2.2
Bocachico <i>Prochilodus magdalenae</i>	93.1 $\pm$ 113.2	2.1
Guacuco <i>Hypostomus hondae</i>	56.0 $\pm$ 55.5	2
Viejita <i>Cyphocharax magdalenae</i>	32.0 $\pm$ 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 5-11. 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  $3 \times 10^6$  (Porcella 1994). Mason et al. (1995) also compared bioaccumulation of inorganic mercury and methylmercury and found that passive uptake of the mercury complexes ( $\text{HgCl}_2$  and  $\text{CH}_3\text{HgCl}$ ) 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

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membrane-bound) and methylmercury (which accumulated in the cytoplasm) led to a greater assimilation of methylmercury during zooplankton grazing.

**Table 5-11. 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 <sup>5</sup>	10 <sup>5.7</sup>	15
Zooplankton	10 <sup>5.5</sup>	10 <sup>5.9</sup>	30
Fish	10 <sup>6.5</sup>	10 <sup>5</sup>	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, *Thalassiosira 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  $\mu\text{g g}^{-1}$  dry weight in the plant/ $\mu\text{g mL}^{-1}$  water day<sup>-1</sup>.

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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 Kalac 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.

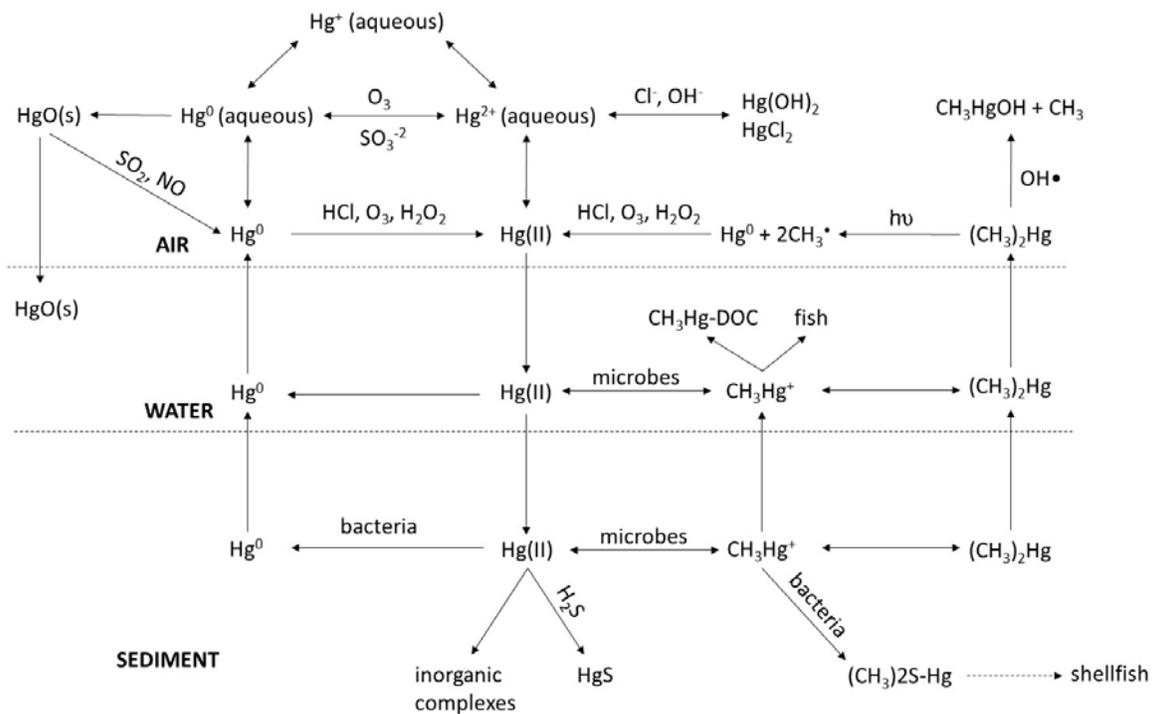
Earthworms, *Lumbricus sp.*, bioaccumulate mercury under laboratory and field conditions in amounts that are dependent on soil concentrations and exposure duration (Cocking et al. 1994). Maximum mercury tissue concentrations in laboratory cultures were only 20% of the 10–14.8 µg/g (ppm) (dry weight) 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).

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## 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.

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**Air.** The primary form of atmospheric mercury, metallic mercury vapor ( $\text{Hg}^0$ ), is oxidized by ozone to other forms (e.g.,  $\text{Hg}^{+2}$ ) and is removed from the atmosphere by precipitation (Brosset and Lord 1991). The oxidation/reduction of mercury with dissolved ozone, hydrogen peroxide, hypochlorite entities, or organoperoxy compounds or radicals may also occur in the atmosphere (Schroeder et al. 1991). 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 ( $\text{Hg}[\text{OH}]^2$ ), 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). 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  $\text{Hg}(\text{II})$  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), 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

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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 ( $11 \times 10^{-17}$  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).

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**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 ( $K_m = 0.160 \text{ day}^{-1}$ ), and the lowest rate constant was observed in non-vegetated sediments at the Alcochete site in Tagus ( $K_m = 0.009 \text{ day}^{-1}$ ). 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

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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  $\text{Hg}^0$  or oxidation yielding  $\text{Hg}^{2+}$  and  $\text{CO}_2$ . 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  $\text{Hg}^{+2}$  to mercuric sulfide by sulfate-reducing bacteria under anaerobic conditions (Revis et al. 1989, 1990). Mercuric sulfide has very limited water solubility ( $4.5 \times 10^{-24}$  mol/L), and thus, in the absence of other solvents, is likely to have limited mobility in soil. Aerobic microorganisms can solubilize  $\text{Hg}^{+2}$  from mercuric sulfide by oxidizing the sulfide through sulfite to sulfate, with the  $\text{Hg}^{+2}$  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  $0.07 \pm 0.15$  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 detection methods. In reviewing data on mercury levels monitored or estimated in

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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 5-12 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 5-13.

**Table 5-12. Lowest Limit of Detection Based on Standards<sup>a,b</sup>**

Media	Detection limit	Reference
Air	30 pg/m <sup>3</sup> (particulate); 45 pg/m <sup>3</sup> (vapor)	EPA 1999a Method IO-5
Drinking water	0.2 ng/L	EPA 2002 Method 1631E
Surface water and groundwater	0.2 ng/L	EPA 2002 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 2019

<sup>a</sup>Detection limits based on using appropriate preparation and analytics. These limits may not be possible in all situations.

<sup>b</sup>A review of analytical methods to detect mercury in environmental matrices has been published by Suvarapu and Baek (2017).

**Table 5-13. Summary of Environmental Levels of Mercury Worldwide**

Media	Low	High	For more information
Outdoor air (ng/m <sup>3</sup> )	0.000161 (particulate)	174,000 (vapor)	Section 5.5.1
Indoor air (ng/m <sup>3</sup> )	<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 5-14.

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**Table 5-14. Mercury Levels in Water, Soil, and Air at National Priorities List (NPL) Sites**

Medium	Median <sup>a</sup>	Geometric mean <sup>a</sup>	Geometric standard deviation <sup>a</sup>	Number of quantitative measurements	NPL sites
Water (ppb)	2	3.50	16.3	316	188
Soil (ppb)	1,750	2,720	28.7	376	216
Air (ppbv)	0.334	0.382	61.2	25	17

<sup>a</sup>Concentrations found in ATSDR site documents from 1981 to 2019 for 1,867 NPL sites (ATSDR 2019). 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<sup>0</sup> (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<sup>3</sup>, although levels can vary due to local sources (Gworek et al. 2017). 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<sup>3</sup>, with a maximum level of 248 ng/m<sup>3</sup> 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<sup>3</sup>. The arithmetic mean mercury levels associated with PM<sub>10</sub> (particulate matter 10-micron diameter or less) ranged from 0.00619 to 1.50 ng/m<sup>3</sup>, while the arithmetic means for mercury associated with PM<sub>2.5</sub> were 0.000161–0.000317 ng/m<sup>3</sup>.

The Atmospheric Mercury Network (AMN) and the Mercury Deposition Network 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 2008 to 2018 are shown in Table 5-15.

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**Table 5-15. Atmospheric Mercury Network (AMN) Average Gaseous Elemental Mercury Concentrations, 2008–2018**

Year	Averaged gaseous elemental mercury level (ng/m <sup>3</sup> )
2008	1.53
2009	1.48
2010	1.50
2011	1.47
2012	1.49
2013	1.50
2014	1.53
2015	1.47
2016	1.39
2017	1.39
2018	1.33

Source: NADP 2020

As mercury is subject to long-range transport, it has been detected in Arctic atmospheric samples. The median ( $\pm$ SD) concentrations of mercury at Alert, Canada in the fall, winter, spring, and summer were 1.49 ( $\pm$ 0.11), 1.59 ( $\pm$ 0.17), 1.24 ( $\pm$ 0.53), and 1.80 ( $\pm$ 0.35) ng/m<sup>3</sup>, respectively (Kirk et al. 2012).

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<sup>3</sup>, respectively (Brosset and Lord 1991). Higher levels (10–15  $\mu$ g/m<sup>3</sup>) 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<sup>3</sup> (Wiener et al. 1990). Mercury levels ranged from 6.3 to 16.0 ng/m<sup>3</sup> 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<sup>3</sup> in 1988–1989, while the particle-associated aerosol mercury concentration was determined to be 0.03 ng/m<sup>3</sup>, 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<sup>3</sup> and particulate mercury at 0.002–0.06 ng/m<sup>3</sup> 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<sup>3</sup> (Glass et al. 1990).

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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<sup>3</sup> (mean 1.64 ng/m<sup>3</sup>). 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<sup>3</sup>) than measurements at an Atlantic coastal site (1.8 ng/m<sup>3</sup>). The mean concentrations of particle-phase mercury collected at the inland sites (51 and 49 pg/m<sup>3</sup>) were 50% higher than those at the coastal site (34 pg/m<sup>3</sup>). 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<sup>3</sup> 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<sup>3</sup>). Furthermore, a diurnal pattern was observed in the vapor-phase mercury levels measured at the Chicago site. The average concentration (ng/m<sup>3</sup>) 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<sup>3</sup>). Particulate-phase mercury concentrations were also higher at the Chicago site than at the South Haven site (98 versus 19 pg/m<sup>3</sup>). 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<sup>3</sup>; range 1.2–4.2 ng/m<sup>3</sup>), and the concentrations were consistent across all seasons. Particulate-phase mercury concentrations averaged 11 pg/m<sup>3</sup>, 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 µg/m<sup>3</sup>) was found to be significantly higher ( $p < 0.0001$ ) in the exposed homes (range: not detectable to 1.5 µg/m<sup>3</sup>) than in the unexposed homes (range: not detectable to 0.3 µg/m<sup>3</sup>). Among the exposed

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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  $\mu\text{g}/\text{m}^3$  (range: not detectable to 1  $\mu\text{g}/\text{m}^3$ ). Six exposed homes had air mercury concentrations >0.5  $\mu\text{g}/\text{m}^3$ . 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  $\mu\text{g}/\text{m}^3$  (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  $\mu\text{g}/\text{m}^3$ . These values decreased to 0.006–0.071  $\mu\text{g}/\text{m}^3$  by 1990, while background levels near the facility remained at 0.006  $\mu\text{g}/\text{m}^3$ . 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  $\text{mg}/\text{m}^3$ ) with the exception of one station. Mean mercury levels at the control site ranged from 5 to 6  $\mu\text{g}/\text{m}^3$ , while levels at the on-site stations ranged from 6–11, 11–143, 68–174, 71–109, and 4–46  $\mu\text{g}/\text{m}^3$ , depending on the station. Mean particulate mercury levels were 0.00003  $\mu\text{g}/\text{m}^3$  at the control site, compared with mean concentrations at the on-site stations ranging from 0.00006 to 0.00024  $\mu\text{g}/\text{m}^3$  (Turner and Bogle 1993).

### 5.5.2 Water

Concentrations of mercury in rainwater and fresh snow are generally <200 ng/L (ppt) (EPA 1984). 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 and temporal trends for decreasing and increasing concentrations, depending on location (Risch et al. 2012). Fitzgerald et al. (1991)

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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  $\mu\text{g}/\text{m}^2$  (Glass et al. 1991). Antarctic surface snow contained a mean mercury concentration of <1 pg/g (ppt) (Dick et al. 1990). 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  $\mu\text{g}/\text{m}^2$ . 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).

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. Methylated mercury averaged 0.024 ( $\pm 0.009$  SD) ng/L in Canadian Arctic waters and 0.023 ng/L ( $\pm 0.011$  SD) in the Hudson Bay (Kirk et al. 2012). Freshwaters without known sources of mercury contamination generally contain <5 ng/L (ppt) of total mercury in aerobic surface waters (Gilmour and Henry 1991). The USGS conducted sampling from 2012 to 2015 at six locations of the Brownlee Reservoir, Boise, and Snake River in Idaho (MacCoy and Williams 2016). Mercury levels in the water ranged from 0.48 to 8.8 ng/L, with the highest concentration observed in the Brownlee Reservoir. Mercury levels in water-borne particulates in the St. Louis River estuary ranged from 18 to 500 ng/L (ppt) (Glass et al. 1990). Water samples from lakes and rivers in the Ottawa, Ontario, region of Canada had total mercury concentrations of 3.5–11.4 ng/L (ppt), with organic mercury constituting 22–37% of the total mercury (Schintu et al. 1989). Mercury was detected in water samples from Crab Orchard Lake, Illinois, at 70–281 ng/L (ppt) (Kohler et al. 1990). Total mercury concentrations in surface waters of California lakes and rivers ranged from 0.5 to 104.3 ng/L (ppt), with the dissolved particulate fraction being dominant (89%; 0.4–12 ng/L [ppt]) (Gill and Bruland 1990).

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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 Babiartz 1992). Mercury was found at levels  $>0.5 \mu\text{g/L}$  (ppb) in 15–30% of wells tested in some groundwater surveys (EPA 1985). Drinking water is generally assumed to contain  $<0.025 \mu\text{g/L}$  (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  $\mu\text{g/kg}$  (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. 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).

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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.

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 associated with contamination in northeastern Louisiana from mercury-charged manometers 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 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). 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

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proximal to the former plant site and contained about 68 times (4.31 ppm) the regional background concentration.

The top 15 cm of sediments in Wisconsin lakes contained higher levels of mercury (0.09–0.24  $\mu\text{g/g}$  [ppm]) than sediments at lower sediment levels (0.04–0.07  $\mu\text{g/g}$  [ppm]). Because the lakes are not known to receive any direct deposition of mercury, it was postulated that the primary mercury source was atmospheric deposition (Rada et al. 1989). Mercury levels in surface sediments of the St. Louis River ranged from 18 to 500 ng/L (ppt) (Glass et al. 1990). Mercury was detected in sediment samples from Crab Orchard Lake in Illinois at concentrations  $>60 \mu\text{g/L}$  (ppb) (Kohler et al. 1990). Surficial sediment samples from several sites along the Upper Connecting Channels of the Great Lakes in 1985 had mercury concentrations ranging from below the detection limit to 55.80  $\mu\text{g/g}$  (ppm) (mean concentrations ranged from 0.05 to 1.61  $\mu\text{g/g}$  [ppm] at four sites) (Nichols et al. 1991). Mercury concentrations were correlated with particle size fractions and organic matter content (Mudroch and Hill 1989). Surface sediment samples from the Lake Roosevelt/Upper Columbia River in Washington State were found to contain up to 2.7  $\mu\text{g/g}$  (ppm) mercury (Johnson et al. 1990). Mercury concentrations in sediments up to 28 cm in depth in lakes adjacent to coal-fired power plants near Houston, Texas ranged from 255 to 360 mg/kg (ppm) in the summer and from 190 to 279 mg/kg (ppm) in the winter (Wilson and Mitchell 1991).

Surface sediments taken from Canadian lakes receiving atmospheric input from smelters contained between 0.03 and 9.22  $\mu\text{g/g}$  (ppm) mercury, with the highest values being found in lakes nearest the smelters. However, sediment concentrations were not correlated with mercury concentrations in fish from the lakes; the fish concentrations ranged from 0.003 to 0.88  $\mu\text{g/g}$  (ppm), with the highest concentration found in fish from one of the least contaminated lakes (Harrison and Klaverkamp 1990).

Estuarine and coastal marine sediment samples analyzed for the National Oceanic and Atmospheric Administration's (NOAA) National Status and Trends Program between 1984 and 1987 showed that 38 of 175 sites contained mercury concentrations in excess of 0.41  $\mu\text{g/g}$  (ppm) (dry weight) (O'Connor and Ehler 1991). In addition, mercury sediment concentrations at six sites exceeded the NOAA effects range median concentration of 1.3 ppm (dry weight), which is the concentration determined to be equivalent to the median (50<sup>th</sup> percentile) for all sites monitored. These six sites included five sites in the Hudson River/Raritan Estuary, New York Bight, and Raritan Bay areas between New York and New Jersey (ranging from 1.6 to 3.3 ppm dry weight) and one site in the Oakland Estuary in California (2.3 ppm dry weight) (NOAA 1991). Sediments taken from coastal areas off British Columbia, Canada contained concentrations of mercury ranging from 0.05 to 0.20  $\mu\text{g/g}$  (ppm), while mercury concentrations

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in fish from these waters were only slightly higher; BCFs ranged from <1 to 14 (Harding and Goyette 1989).

#### 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).

The most recent 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 5-16.

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

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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).

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.

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

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**Table 5-16. 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

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**Table 5-16. 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

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**Table 5-16. 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

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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 5-17.

**Table 5-17. Trends in Mercury Concentrations Based on Fish Data Aggregated by State from 1988 to 2005<sup>a</sup>**

State	Sites	Number	Species	Begin year	End year	Median mercury (mg/kg)	p-Value	Percent change	Trend
<b>Southeastern United States</b>									
<u>Georgia</u>	<u>112</u>	<u>266</u>	<u>LMB</u>	<u>1991</u>	<u>2001</u>	<u>0.24</u>	<u>&lt;0.001</u>	<u>7.43</u>	<u>Up</u>
Georgia	56	105	CCF	1991	2001	0.10	0.063	6.76	None
<u>Louisiana</u>	<u>324</u>	<u>1,049</u>	<u>LMB</u>	<u>1994</u>	<u>2005</u>	<u>0.39</u>	<u>&lt;0.001</u>	<u>3.78</u>	<u>Up</u>
Louisiana	113	168	CCF	1994	2004	0.10	0.007	11.2	Up
<u>Louisiana</u>	<u>178</u>	<u>328</u>	<u>FD</u>	<u>1994</u>	<u>2004</u>	<u>0.37</u>	<u>0.013</u>	<u>6.23</u>	<u>Up</u>
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
<b>Louisiana</b>	<b>169</b>	<b>444</b>	<b>B</b>	<b>1994</b>	<b>2004</b>	<b>0.52</b>	<b>0.019</b>	<b>-3.26</b>	<b>Down</b>
<i>Louisiana</i>	<i>42</i>	<i>66</i>	<i>BMBU</i>	<i>1995</i>	<i>2004</i>	<i>0.28</i>	<i>0.045</i>	<i>-6.66</i>	<i>Down</i>
North Carolina	37	61	BG	1989	1999	0.10	0.771	0.76	None
<b>South Carolina</b>	<b>129</b>	<b>963</b>	<b>B</b>	<b>1993</b>	<b>2004</b>	<b>0.80</b>	<b>0.003</b>	<b>3.02</b>	<b>Up</b>
South Carolina	70	194	CCF	1994	2004	<0.25	0.277	-2.29	None
<b>South Carolina</b>	<b>188</b>	<b>1,556</b>	<b>LMB</b>	<b>1993</b>	<b>2004</b>	<b>0.38</b>	<b>&lt;0.001</b>	<b>-3.13</b>	<b>Down</b>
<b>Midwestern United States</b>									
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

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**Table 5-17. Trends in Mercury Concentrations Based on Fish Data Aggregated by State from 1988 to 2005<sup>a</sup>**

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

<sup>a</sup>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 (MacCoy and Williams 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.

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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 5-18.

**Table 5-18. 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	<i>Kyphosus sectatrix</i>	78.64	44.87	135.6	57.1
Bicolor damsel	<i>Stegastes partitus</i>	26.60	2.96	29.55	11.1
Black grouper	<i>Mycteroperca bonaci</i>	347.2	85.95	445.2	24.8
Blue runner	<i>Caranx crysos</i>	40.95	NR	NR	NR
Blue stripe grunt	<i>Haemulon sciurus</i>	390.9	108.8	506.1	27.8
Blue tang	<i>Acanthurus coeruleus</i>	35.28	5.32	44.08	15.1
Brown chromis damsel	<i>Chromis multilineata</i>	113.0	NR	NR	NR
Doctorfish tang	<i>Acanthurus chirurgus</i>	52.97	7.58	60.98	14.3
French angelfish	<i>Pomacanthus paru</i>	43.42	22.11	65.53	50.9
Gray angelfish	<i>Pomacanthus arcuatus</i>	58.11	26.34	91.63	45.3
Graysby grouper	<i>Cephalopholis cruentata</i>	322.0	68.33	417.4	21.2
Great barracuda	<i>Sphyaena barracuda</i>	1,713.8	882.3	3,401.4	51.5
Hogfish	<i>Lachnolaimus maximus</i>	129.6	41.44	200.0	31.9
Knobbed porgy	<i>Calamus nodosus</i>	153.0	50.23	203.2	32.8
Porkfish	<i>Anisotremus virginicus</i>	1,555.1	1,970.9	6,842.3	126.7
Princess parrotfish	<i>Scarus taeniopterus</i>	66.84	NR	NR	NR
Red lionfish	<i>Pterois volitans</i>	174.9	44.75	225.8	1.7
Redband parrotfish	<i>Sparisoma aurofrenatum</i>	42.52	10.17	56.91	23.9
Rock Beauty angelfish	<i>Holacanthus tricolor</i>	20.53	1.52	22.04	7.4
Saucereye porgy	<i>Calamus</i>	114.6	NR	NR	NR
Schoolmaster snapper	<i>Lutjanus apodus</i>	143.1	NR	NR	NR
Scrawled filefish	<i>Aluterus scriptus</i>	76.79	6.24	82.57	8.1
Sergeant major damsel	<i>Abudefduf saxatilis</i>	57.19	NR	NR	NR
Spanish grunt	<i>Haemulon macrostomum</i>	505.9	25.55	531.5	5.0

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**Table 5-18. 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	<i>Bodianus rufus</i>	300.8	95.90	392.1	31.9
Spotted goatfish	<i>Pseudupeneus maculatus</i>	102.2	48.03	184.1	46.9
Stoplight parrotfish	<i>Sparisoma viride</i>	30.22	3.33	32.58	11.0
Trumpetfish	<i>Aulostomus maculatus</i>	133.3	NR	NR	NR
White grunt	<i>Haemulon plumierii</i>	304.1	53.05	340.1	10.6
Yellowtail damsel	<i>Chrysiptera parasema</i>	35.91			
Yellowtail snapper	<i>Ocyurus chrysurus</i>	104.4	45.45	149.8	43.5
<b>Long Key Hard Bottom</b>					
Atlantic spadefish	<i>Chaetodipterus faber</i>	242.6	66.47	310.9	27.4
Bermuda chub	<i>Kyphosus sectatrix</i>	50.99	23.80	97.41	46.7
Black grouper	<i>Mycteroperca bonaci</i>	343.1	13.14	356.3	3.8
Blue runner	<i>Caranx crysos</i>	252.2	166.0	460.7	65.8
Blue stripe grunt	<i>Haemulon sciurus</i>	265.4	65.28	356.1	24.6
Blue tang	<i>Acanthurus coeruleus</i>	40.90	NR	NR	NR
Cocoa damsel	<i>Stegastes variabilis</i>	118.4	NR	NR	NR
Doctorfish tang	<i>Acanthurus chirurgus</i>	96.37	NR	NR	NR
French angelfish	<i>Pomacanthus paru</i>	53.31	NR	NR	NR
Gray angelfish	<i>Pomacanthus arcuatus</i>	17.49	0.16	17.65	0.9
Gray snapper	<i>Lutjanus griseus</i>	190.2	31.10	216.1	16.4
Graysby grouper	<i>Cephalopholis cruentata</i>	152.5	NR	NR	NR
Great barracuda	<i>Sphyrnaena barracuda</i>	3,317.5	NR	NR	NR
Hogfish	<i>Lachnolaimus maximus</i>	142.3	55.66	257.5	39.1
Horse-eye jack	<i>Caranx latus</i>	147.8	NR	NR	NR
Lane snapper	<i>Lutjanus synagris</i>	256.2	NR	NR	NR
Leatherjacket	<i>Oligoplites saurus</i>	747.7	43.99	791.7	5.9
Lookdown	<i>Selene vomer</i>	234.9	90.64	322.3	38.5
Mutton snapper	<i>Lutjanus analis</i>	196.9	NR	NR	NR
Ocean surgeon	<i>Acanthurus bahianus</i>	41.41	0.02	41.43	0.0
Pigfish	<i>Orthopristis chrysoptera</i>	857.5	NR	NR	NR
Planehead filefish	<i>Stephanolepis hispidus</i>	77.57	NR	NR	NR
Porkfish	<i>Anisotremus virginicus</i>	501.6	151.1	702.5	30.1
Red grouper	<i>Epinephelus morio</i>	197.8	24.77	232.8	12.5
Red lionfish	<i>Pterois volitans</i>	239.3	25.01	264.4	10.4
Rock hind grouper	<i>Epinephelus adscensionis</i>	144.0	9.35	153.4	6.5

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**Table 5-18. 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	<i>Haemulon plumierii</i>	320.5	78.05	494.7	24.3
Yellow jack	<i>Carangoides bartholomaei</i>	236.4	129.6	366.0	54.8
Yellowtail snapper	<i>Ocyurus chrysurus</i>	142.5	38.52	223.2	27.0

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

Source: Rumbold et al. 2018

As part of the National Pesticide Monitoring Program (NPMP), the U.S. Fish and Wildlife Service collected freshwater fish during 1976–1977 from 98 monitoring stations nationwide and analyzed them for mercury and other heavy metals (May and McKinney 1981). As part of this program, duplicate composite samples of a bottom-dwelling species and several representative predatory species were collected. Bottom-dwelling species sampled included common carp, common sucker, and channel catfish or other catfish species. Predatory species sampled were rainbow, brown, brook, or lake trout at cold water stations; largemouth bass or other sunfish family members, such as crappie or bluegill, at warm water stations; and walleye or other perch family members at cool water stations. May and McKinney (1981) reported that the mean concentration of mercury was 0.153 mg/kg (wet weight basis) in the 1972 NPMP survey and that the mean concentration declined significantly to 0.112 mg/kg (range 0.01–0.84 mg/kg) in the 1976–1977 survey. This decline was presumably due to curtailed production, use, and emissions of mercury (Lowe et al. 1985). May and McKinney (1981) identified an arbitrary 85<sup>th</sup> percentile concentration of 0.19 mg/kg for mercury to identify monitoring stations having fish with higher than normal concentrations of mercury. Most of these stations were downstream of industrial sites (e.g., chloralkali operations, pulp and paper plants, or pre-1900 gold and silver mining operations), while others were located in areas with major mercury ore (cinnabar) deposits. In a follow-up NPMP study conducted from 1980 to 1981, Lowe et al. (1985) reported a geometric mean mercury concentration of 0.11 mg/kg (wet weight) (range 0.01–1.10 mg/kg). The study authors reported that the downward trend in mercury residues in fish reported by May and McKinney (1981) may have leveled off, since no significant difference in the geometric mean values was detected in the follow-up study conducted in 1984–1985 as part of the National Contaminant Biomonitoring Program (Lowe et al. 1985; Schmitt and Brumbaugh 1990). However, large variations in mercury uptake among the fish species sampled, as well as among size classes of fish within the same species, may mask actual trends.

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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 1992b). 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 10<sup>th</sup> 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 10<sup>th</sup> percentile (EPA 1992b).

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 5-19 and 5-20, respectively. As Table 5-19 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 5-20 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,

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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 5-19. Mercury Concentrations (mg/kg) for Largemouth Bass Collected in Various States Throughout the United States (1990–1995)**

State	Number of fish	Minimum	Mean <sup>a</sup>	Maximum <sup>b</sup>
Alabama	914	0.100	0.393	<b>1.630</b>
Arizona	35	0.700	<b>1.369</b>	<b>2.620</b>
Arkansas	1,190	0.030	0.675	<b>3.170</b>
California	517	0.030	0.291	<b>1.800</b>
Connecticut	507	0.032	0.505	<b>2.645</b>
District of Columbia	11	0.037	0.153	0.458
Florida	2,000	0.020	0.645	<b>4.360</b>
Georgia	968	0.010	0.274	<b>2.286</b>
Illinois	305	0.010	0.018	0.880
Louisiana	452	0.001	0.391	<b>1.883</b>
Maine	137	0.071	0.634	<b>1.343</b>
Massachusetts	152	0.045	0.399	<b>1.100</b>
Mississippi	505	0.090	0.651	<b>2.630</b>
Missouri	106	0.002	0.257	0.608
Nebraska	182	0.080	0.343	0.920
New Hampshire	35	0.210	0.573	<b>1.400</b>
New Jersey	173	0.030	0.664	<b>8.940</b>
New York	53	0.050	0.462	0.950
North Carolina	1,569	0.020	0.532	<b>3.600</b>
Oregon	140	0.030	0.332	0.980
Pennsylvania	139	0.090	0.293	0.750
South Carolina	505	0.230	0.994	<b>3.330</b>
Tennessee	64	0.100	0.255	0.830
Texas	58	0.043	0.237	0.657
Vermont	1	0.150	0.802	<b>1.200</b>
Washington	20	0.024	0.137	0.350

<sup>a</sup>Weighted average of composite samples where the weight is the number of fish in each composite ( $\sum(C_i \times N_i)/N_t$ , 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).

<sup>b</sup>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

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

State	Number of fish	Minimum	Mean <sup>a</sup>	Maximum <sup>b</sup>
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	<b>1.1143</b>
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	<b>1.800</b>
Ohio	574	0.018	0.118	1.040
Oklahoma	324	0.100	0.193	0.640
South Carolina	42	0.250	0.345	<b>1.610</b>
Tennessee	138	0.100	0.173	0.650
Texas	44	0.043	0.193	<b>1.186</b>
West Virginia	65	0.030	0.139	<b>1.583</b>

<sup>a</sup>Weighted average of composite samples where the weight is the number of fish in each composite ( $\sum(C_i \times N_i)/N_i$ , where  $C_i$  and  $N_i$  are the concentrations and number of fish in each composite sample, respectively, and  $N_i$  is the total number of fish in all composites).

<sup>b</sup>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 fresh water 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 5-21. This study

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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 5-21. Combined Data on Mercury Concentrations in Selected Fish Species Sampled in the Northeast<sup>a</sup>**

Species	Number of samples <sup>b</sup>	Mean mercury concentration	Minimum–maximum mercury <sup>c</sup> concentration range (mg/kg)
Largemouth bass	1,019	0.51	0– <b>8.94</b>
Smallmouth bass	738	0.53	0.08– <b>5.0</b>
Yellow perch	1,346	0.40	0– <b>3.15</b>
Eastern chain pickerel	157	0.63	0– <b>2.81</b>
Lake trout	877	0.32	0– <b>2.70</b>
Walleye <sup>d</sup>	257	0.77	0.10– <b>2.04</b>
Brown bullhead	421	0.20	0– <b>1.10</b>
Brook trout	200	0.26	0–0.98

<sup>a</sup>Northeastern states include Main, Vermont, New Hampshire, Massachusetts, Rhode Island, Connecticut, New York, and New Jersey.

<sup>b</sup>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.

<sup>c</sup>Maximum tissue concentrations shown in **bold type** exceed the U.S. Food and Drug Administration action level of 1 mg/kg (1 ppm).

<sup>d</sup>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

Lake trout taken from Lake Ontario between 1977 and 1988 did show a progressive decline in mercury contamination from 0.24 mg/kg in 1977 to 0.12 mg/kg in 1988 (Borgmann and Whittle 1991). Samples of zooplankton taken from an Illinois lake in 1986 contained approximately 0.01 mg/kg mercury; however, fish that fed on the zooplankton had whole-body mercury concentrations ranging from 0.0116 mg/kg for inedible shad to 0.069 mg/kg for edible largemouth bass, indicating bioaccumulation was occurring up the aquatic food chain. Older fish generally had higher mercury concentrations (Kohler et al. 1990). Mercury concentrations in crayfish taken from 13 Ontario lakes with no known mercury inputs ranged from 0.02 to 0.64 mg/kg; the concentrations were positively correlated with organism weight and fish mercury concentrations (Allard and Stokes 1989). Brown trout taken from Lake Ontario contained between 0.18 and 0.21 mg/kg mercury in unskinned fillets and between 0.24 and 0.26 mg/kg mercury in skinned fillets, indicating that methylmercury is associated with the protein fraction of fish tissue (Gutenmann and Lisk 1990).

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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).

Typical mercury concentrations in large carnivorous freshwater fish (e.g., pike) and large marine fish (e.g., swordfish, shark, and tuna) have been found to >1 mg/kg (EPA 1984; Fairey et al. 1997; FDA 1994; Hellou et al. 1992; Hueter et al. 1995), with mercury content again being positively correlated with the age of the fish (Gutenmann et al. 1992; Hueter et al. 1995). Methylmercury concentrations in muscle tissue of nine species of sharks were analyzed from four locations off Florida (Hueter et al. 1995). Muscle tissue methylmercury concentration averaged 0.88 mg/kg (wet weight) and ranged from 0.06 to 2.87 mg/kg, with 33.1% of the samples exceeding the FDA action level (1 ppm [1 mg/kg]). A positive correlation between methylmercury and shark body length (size) also was found, such that sharks larger than 200 cm in total length contained methylmercury concentrations >1 mg/kg. Sharks collected off the southern and southwestern coastal areas contained significantly higher concentrations than those caught in the northeast coastal region (Cape Canaveral and north).

Methylmercury concentrations were highest in the Caribbean reef shark (*Carcharhinus perezi*). The two most abundant shark species in the U.S. east coast commercial shark fishery, sandbar (*C. plumbeus*) and blacktip (*C. limbatus*) sharks, are of special concern with respect to human health. Although the mean concentration of methylmercury in the sandbar shark (0.77 mg/kg) was below the average for all sharks, sandbar shark tissues contained up to 2.87 ppm methylmercury, and 20.9% of the samples exceeded the FDA action level of 1 ppm (1 mg/kg). A total of 71.4% of the blacktip shark samples (mean, 1.3 mg/kg) exceeded the FDA action level. The study authors suggested that continued monitoring of methylmercury concentrations in various sharks species in the commercial marketplace is warranted. 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

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(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).

As part of the NOAA Status and Trends Program conducted from 1984 to 1987, mercury concentrations were analyzed in four marine bivalve species in U.S. coastal waters (NOAA 1987). Mercury concentrations in bivalve tissues were 0.01–0.48 mg/kg dry weight in oysters (*Crassostrea virginica*), 0.28–0.41 mg/kg in the Hawaiian oyster (*Ostrea sandwichensis*), 0.05–0.47 mg/kg in the blue mussel (*Mytilus edulis*), and 0.04–0.26 mg/kg in the California mussel (*Mytilus californianus*). Oysters (*Crassostrea virginica*) collected around the Gulf of Mexico between 1986 and 1989 had mercury concentrations ranging from <0.01 to 0.72 mg/kg (mean 0.127 mg/kg) (Presley et al. 1990). Oysters taken from the Mississippi Sound in 1986 generally did not contain mercury at levels exceeding the detection limit (0.02 mg/kg), although two samples had detectable mercury levels of 0.66 and 6.6 mg/kg (Lytle and Lytle 1990).

**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 walrus 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 (see Table 5-22). 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.

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**Table 5-22. 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)	
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

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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). Although data on mercury distribution among freshwater vascular plant parts is lacking for unpolluted systems, 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 ( $\text{HgCl}_2$ ) than organic methylmercury ( $\text{CH}_3\text{HgCl}$ ) 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 ( $\text{HgF}_2$  and  $\text{HgCl}_2$ ) 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.

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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 (Kalac 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).

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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

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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.*** 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. 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). Vietnamese and Indian people may also use mercury as alternative medicine. 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, blood mercury 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

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becoming weak; exhibiting a rash; and swollen, red, painful hands and feet. Testing indicated diminished reflexes with blood mercury levels of 176 and 209  $\mu\text{mol/L}$  (Weinstein and Bernstein 2003).

In the United States, people may obtain metallic mercury (sometimes under the name azogue) in shops called botanicas. These stores typically stock medicinal plants, traditional medicines, incense, candles, creams, and perfumes. 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 (NDJEP 2007; Riley et al. 2006). 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 (Bloom 1992; Davis et al. 2014; De Winter-Sorkina et al. 2003; EFSA 2012; EPA 1999b; Kim et al. 2016b; Lescord et al. 2018; Mahaffey et al. 2004; Nielsen et al. 2015; Sirot et al. 2008; WHO 1990; You et al. 2014, 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 (see Section 3.1 Toxicokinetics).

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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; 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<sup>0</sup> vapor and particulate Hg<sup>0</sup> 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<sup>0</sup> 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

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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 2019) (Tables 5-23–5-32). Based on survey data for the period 2015–2016 (the most recent data available in CDC 2019), the geometric mean total blood mercury level in the adult U.S. population was estimated to be 0.810 µg/L (95% CI 0.740, 0.886). The geometric mean methylmercury blood level was 0.518 µg/L (95% CI 0.456, 0.588). Total and methylmercury blood levels in young children were lower than in adults. The 50<sup>th</sup> percentiles for total blood mercury levels in children 1–5 years of age were less than the detection limit (0.28 µg/L) in 2015–2016. For the 2011–2012 period, the detection limits for total mercury were lower, reporting a geometric mean total blood mercury level of 0.262 µg/L (95% CI 0.237, 0.291) in children 1–5 years of age. The 50<sup>th</sup> 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 2015–2016 period, the 50<sup>th</sup> percentiles of total urinary mercury were 0.140 µg/L (95% CI 0.130, 0.150) in adults and below the detection limit (0.13 µg/L) in children 3–5 years of age.

### 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 use mercury as part of spiritual practices (Riley et al. 2001).

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**Table 5-23. 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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
<b>Gender</b>							
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

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**Table 5-23. 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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
<b>Race/ethnicity</b>							
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

<sup>a</sup>The limit 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 2019

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**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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
Total	2011–2012	0.703 (0.0617–0.801)	0.640 (0.0580–0.730)	1.38 (1.14–1.72)	2.87 (2.39–3.62)	4.40 (3.50–5.71)	7,920
	2013–2014	0.683 (0.0621–0.751)	0.620 (0.0540–0.690)	1.29 (1.14–1.46)	2.65 (2.32–3.08)	4.36 (3.65–4.97)	5,215
	2015–2016	0.678 (0.0619–0.743)	0.600 (0.0540–0.690)	1.26 (1.07–1.47)	2.55 (2.17–3.10)	4.25 (3.44–4.94)	4,988
<b>Age group</b>							
1–5 years	2011–2012	0.262 (0.0237–0.291)	0.250 (0.0220–0.270)	0.390 (0.0340–0.450)	0.680 (0.0540–0.880)	0.99 (0.0790–1.21)	713
	2013–2014	NC	<LOD	0.410 (0.0370–0.450)	0.810 (0.0710–0.990)	1.21 (1.05–1.48)	818
	2015–2016	NC	<LOD	0.380 (0.0340–0.430)	0.690 (0.0540–0.830)	1.06 (0.0840–1.36)	790
6–11 years	2011–2012	0.330 (0.0287–0.379)	0.320 (0.0280–0.360)	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 (0.0290–0.340)	0.480 (0.0430–0.570)	0.920 (0.0750–1.13)	1.33 (1.01–2.28)	1,023
12–19 years	2011–2012	0.411 (0.0355–0.476)	0.370 (0.0320–0.450)	0.680 (0.0590–0.800)	1.32 (1.08–1.75)	2.25 (1.46–2.87)	1,129
	2013–2014	0.412 (0.0367–0.463)	0.350 (0.0310–0.420)	0.630 (0.0530–0.750)	1.20 (0.0900–1.67)	1.87 (1.30–2.38)	627
	2015–2016	0.395 (0.0356–0.439)	0.340 (0.0300–0.370)	0.590 (0.0470–0.750)	1.00 (0.0850–1.35)	1.89 (1.02–3.34)	565
≥20 years	2011–2012	0.863 (0.0753–0.990)	0.790 (0.0690–0.940)	1.68 (1.36–2.12)	3.35 (2.71–4.31)	5.02 (3.94–6.96)	5,030
	2013–2014	0.814 (0.0736–0.900)	0.740 (0.0650–0.850)	1.54 (1.36–1.71)	3.08 (2.73–3.56)	4.88 (4.36–5.21)	2,695
	2015–2016	0.810 (0.0740–0.886)	0.740 (0.0660–0.830)	1.47 (1.28–1.75)	2.86 (2.50–3.44)	4.66 (3.91–5.96)	2,610
<b>Gender</b>							
Males	2011–2012	0.712 (0.0623–0.815)	0.650 (0.0570–0.730)	1.40 (1.17–1.72)	3.00 (2.44–3.91)	4.94 (3.50–6.79)	3,968
	2013–2014	0.688 (0.0617–0.767)	0.620 (0.0530–0.720)	1.30 (1.12–1.54)	2.76 (2.36–3.34)	4.52 (3.65–5.23)	2,587
	2015–2016	0.679 (0.0621–0.743)	0.610 (0.0530–0.700)	1.29 (1.06–1.60)	2.45 (2.06–3.31)	4.67 (3.77–5.39)	2,488
Females	2011–2012	0.694 (0.0609–0.791)	0.640 (0.0580–0.740)	1.36 (1.09–1.75)	2.81 (2.28–3.50)	4.03 (3.29–5.08)	3,952
	2013–2014	0.678 (0.0617–0.745)	0.610 (0.0530–0.700)	1.27 (1.14–1.42)	2.56 (2.17–3.08)	4.15 (3.37–4.93)	2,628
	2015–2016	0.677 (0.0608–0.754)	0.600 (0.0530–0.700)	1.23 (1.02–1.42)	2.57 (2.20–3.10)	3.95 (3.10–4.55)	2,500
<b>Race/ethnicity</b>							
Mexican Americans	2011–2012	0.483 (0.0424–0.550)	0.480 (0.0400–0.560)	0.810 (0.0720–0.900)	1.44 (1.16–1.63)	1.90 (1.57–2.19)	1,077
	2013–2014	0.487 (0.0433–0.547)	0.430 (0.0390–0.510)	0.760 (0.0690–0.870)	1.41 (1.14–1.69)	1.98 (1.70–2.38)	969
	2015–2016	0.540 (0.0522–0.559)	0.530 (0.0470–0.570)	0.840 (0.0790–0.920)	1.41 (1.19–1.60)	1.81 (1.54–2.33)	994
Non- Hispanic blacks	2011–2012	0.679 (0.0542–0.852)	0.630 (0.0500–0.790)	1.24 (0.0880–1.72)	2.45 (1.84–3.14)	3.80 (2.70–5.37)	2,195
	2013–2014	0.699 (0.0614–0.796)	0.650 (0.0570–0.750)	1.20 (1.08–1.40)	2.30 (1.65–2.96)	3.34 (2.35–5.93)	1,119
	2015–2016	0.698 (0.0587–0.829)	0.630 (0.0510–0.760)	1.21 (1.04–1.47)	2.49 (2.01–3.51)	4.51 (2.74–5.87)	1,070

## 5. POTENTIAL FOR HUMAN EXPOSURE

**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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
Non-Hispanic whites	2011–2012	0.688 (0.0582–0.813)	0.630 (0.0550–0.750)	1.38 (1.09–1.82)	2.83 (2.18–3.82)	4.25 (3.02–6.24)	2,493
	2013–2014	0.672 (0.0598–0.755)	0.620 (0.0520–0.720)	1.30 (1.12–1.51)	2.61 (2.18–3.08)	4.15 (3.35–4.98)	1,848
	2015–2016	0.638 (0.0563–0.723)	0.570 (0.0500–0.670)	1.22 (0.0940–1.47)	2.29 (1.82–2.72)	3.95 (2.56–4.55)	1,511
All Hispanics	2011–2012	0.612 (0.0527–0.710)	0.590 (0.0490–0.700)	1.08 (0.0890–1.33)	1.96 (1.60–2.68)	3.03 (2.37–3.86)	1,931
	2013–2014	0.551 (0.0486–0.624)	0.490 (0.0420–0.580)	0.910 (0.0820–1.10)	1.76 (1.44–2.12)	2.59 (2.06–3.14)	1,481
	2015–2016	0.607 (0.0546–0.675)	0.570 (0.0520–0.640)	1.00 (0.0870–1.17)	1.75 (1.38–2.33)	2.60 (1.85–3.25)	1,664
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

<sup>a</sup>The limit of detection for survey years 2011–2012, 2013–2014, and 2015–2016 were 0.16, 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-25. 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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
<b>Gender</b>							
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-25. 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**

Race/ethnicity	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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

<sup>a</sup>The limit 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
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
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
≥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
<b>Gender</b>							
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
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
<b>Race/ethnicity</b>							
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
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
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
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

<sup>a</sup>The limit of detection for survey years 2011–2012, 2013–2014, and 2015–2016 were 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-27. 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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
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
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
≥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
<b>Gender</b>							
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
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
<b>Race/ethnicity</b>							
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
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-27. 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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
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
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

<sup>a</sup>The limit of detection for survey years 2011–2012, 2013–2014, and 2015–2016 were 0.16, 0.16, and 0.16 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-28. 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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
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
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)	1,121
	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
≥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
<b>Gender</b>							
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
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
<b>Race/ethnicity</b>							
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
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-28. 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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
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
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

<sup>a</sup>The limit of detection for survey years 2011–2012, 2013–2014, and 2015–2016 were 0.12, 0.12, and 0.12 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-29. 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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
<b>Gender</b>							
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
<b>Race/ethnicity</b>							
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-29. 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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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

<sup>a</sup>The limit 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
3–5 years	2015–2016	NC	<LOD	<LOD	0.160 (<LOD–0.240)	0.280 (0.190–0.510)	496
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
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
≥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
<b>Gender</b>							
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
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
<b>Race/ethnicity</b>							
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
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**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) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
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
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

<sup>a</sup>The limit of detection for survey years 2011–2012, 2013–2014, and 2015–2016 were 0.05, 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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-31. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations ( $\mu\text{g/g}$  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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
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
<b>Gender</b>							
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
<b>Race/ethnicity</b>							
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

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-31. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations ( $\mu\text{g/g}$  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) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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

<sup>a</sup>The limit of detection was based on non-creatinine corrected urinary concentrations, which were 0.14, 0.11, 0.08, and 0.08  $\mu\text{g/L}$  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 2019

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-32. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations ( $\mu\text{g/g}$  of Creatinine) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
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
<b>Age group</b>							
3–5 years	2015–2016	NC	<LOD	<LOD	0.667 (<LOD–0.818)	0.994 (0.818–1.13)	495
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
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
≥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
<b>Gender</b>							
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
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
<b>Race/ethnicity</b>							
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)	.692 (0.530–0.810)	1.04 (0.791–1.30)	585
Non-Hispanic blacks	2011–2012	0.280 (0.245–0.320)	0.261 (0.224–0.294)	0.467 (0.411–0.529)	.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)	.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)	.642 (0.450–0.848)	0.901 (0.695–1.18)	674

## 5. POTENTIAL FOR HUMAN EXPOSURE

**Table 5-32. Geometric Mean and Selected Percentiles of Total Mercury Creatinine Corrected Urinary Concentrations ( $\mu\text{g/g}$  of Creatinine) for the U.S. Population from the National Health and Nutrition Examination Survey (NHANES) 2011–2016**

	Survey years	Geometric mean (95% CI)	Selected percentiles (95% CI) <sup>a</sup>				Sample size
			50 <sup>th</sup>	75 <sup>th</sup>	90 <sup>th</sup>	95 <sup>th</sup>	
Non-	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
Hispanic	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
whites	2015–2016	NC	<LOD	0.349 (0.300–0.400)	0.708 (0.563–0.886)	1.10 (0.827–1.32)	932
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
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

<sup>a</sup>The limit of detection was based on non-creatinine corrected urinary concentrations, which were 0.05, 0.13, and 0.13  $\mu\text{g/L}$  for survey years 2011–2012, 2013–2014, and 2015–2016, 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 2019