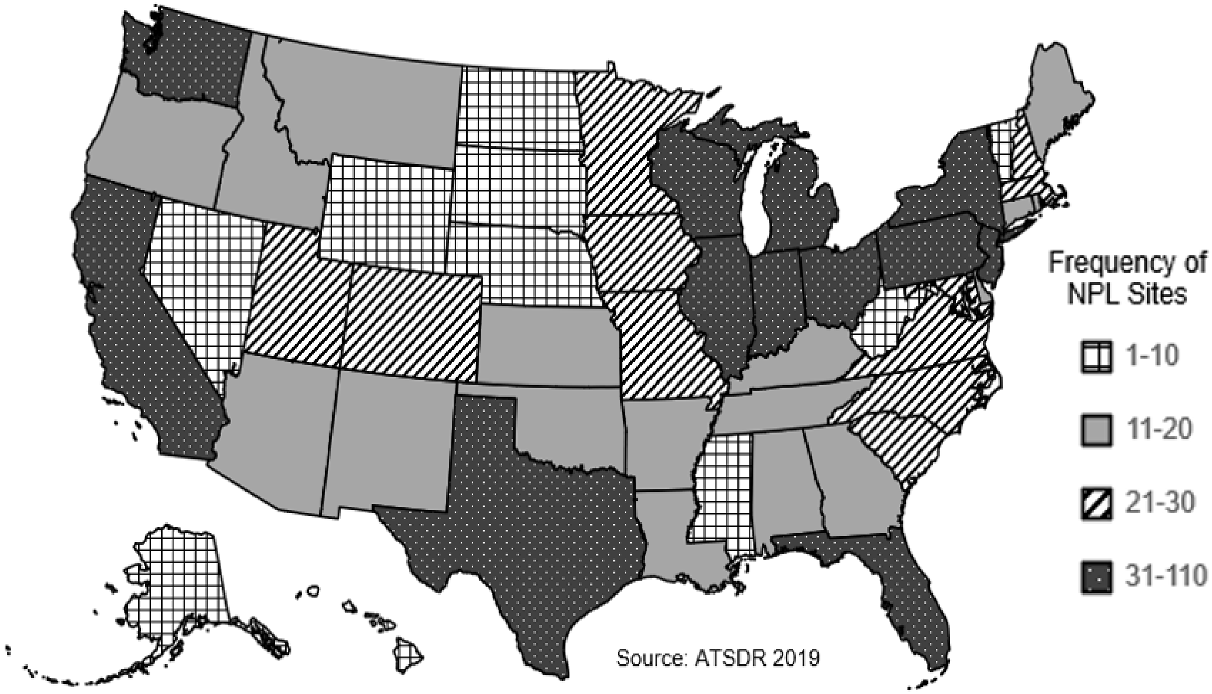
CHAPTER 5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

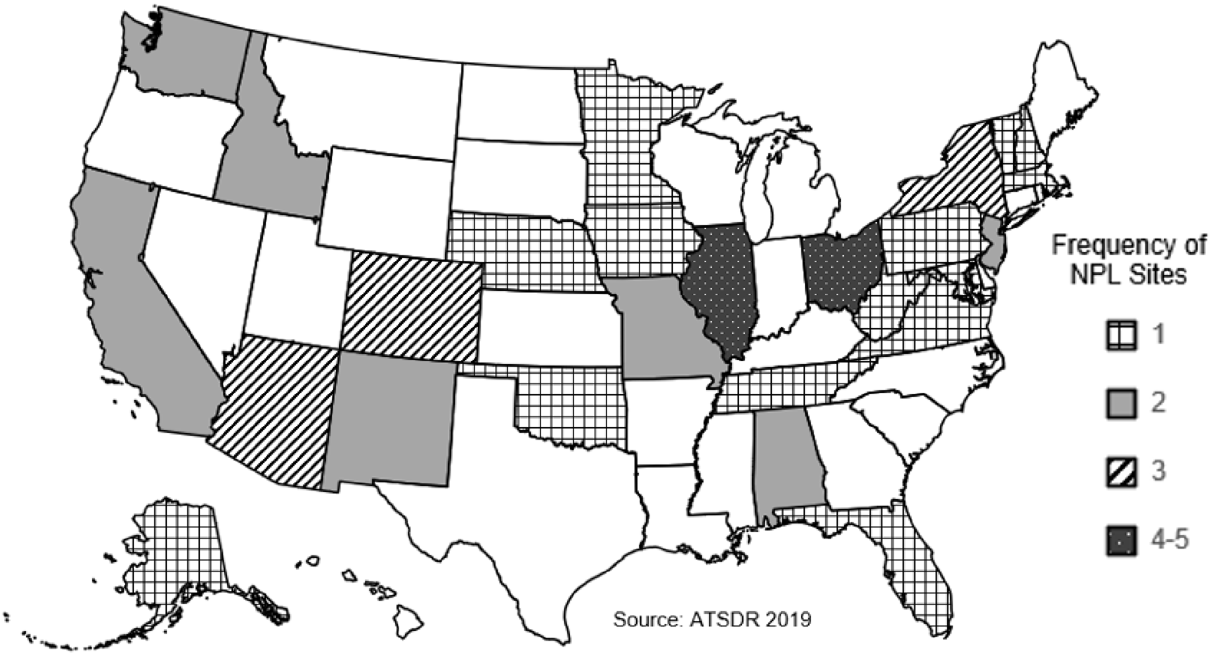
Pb and Pb compounds have been identified in at least 1,287 and 46 sites, respectively, of the 1,867 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (ATSDR 2019). However, the number of sites evaluated for Pb is not known. The number of sites in each state is shown in Figures 5-1 and 5-2, respectively. Of these 1,287 sites for Pb, 1,273 are located within the United States, 2 are located in the Virgin Islands, 2 are located in Guam, and 10 are located in Puerto Rico (not shown). All the sites for Pb compounds are only in the United States.

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



Source: ATSDR 2019

Figure 5-2. Number of NPL Sites with Lead Compound Contamination



- Pb is an element found in concentrated and easily accessible Pb ore deposits that are widely distributed throughout the world.
- The general population may be exposed to Pb in ambient air, foods, drinking water, soil, and dust. For adults, exposure to levels of Pb beyond background are usually associated with occupational exposures.
- For children, exposure to high levels of Pb are associated with living in areas contaminated by Pb (e.g., soil or indoor dust in older homes with Pb paint). Exposure usually occurs by hand-to-mouth activities.
- As an element, Pb does not degrade. However, particulate matter contaminated with Pb can move through air, water, and soil.
- Atmospheric deposition is the largest source of Pb found in soils. Pb is transferred continuously between air, water, and soil by natural chemical and physical processes such as weathering, runoff, precipitation, dry deposition of dust, and stream/river flow; however, soil and sediments appear to be important sinks for Pb.
- Pb adsorbs strongly to most soils, which limits the rate of leaching of Pb from soil. Soil acidity (pH) is the most important factor affecting solubility, mobility, and phytoavailability of Pb in soil.

> Other conditions that increase Pb mobility in soils are reducing conditions (low redox potential; for example, anoxia) and high chloride content.

Pb is dispersed throughout the environment primarily as the result of anthropogenic activities. In the air, Pb is in the form of particles and is removed by rain or gravitational settling. The solubility of Pb compounds in water is a function of pH, ionic strength, and the presence of humic material. Solubility is highest in acidic water. Soil and sediment are an important sink for Pb. Because Pb is strongly adsorbed to soil, very little is transported through runoff to surface water or leached to groundwater except under acidic conditions. Anthropogenic sources of Pb include the mining and smelting of ore, manufacture and use of Pb-containing products, combustion of coal and oil, and waste incineration. Many anthropogenic sources of Pb, most notably leaded gasoline, Pb-based paint, Pb solder in food cans, Pb-arsenate pesticides, and shot and sinkers, have been eliminated or are regulated. Pb compounds released to the environment may be transformed to other Pb compounds; however, Pb is an element and cannot be destroyed or degraded. Because Pb does not degrade over time, deposits of Pb in the environment by current and former uses leave their legacy as higher concentrations of Pb in the environment. These deposits can continue to be a source for potential Pb exposure (e.g., soil particles containing Pb also may be resuspended and redeposited). Plants and animals may bioconcentrate Pb, but Pb is not biomagnified in the aquatic or terrestrial food chain.

The general population may be exposed to Pb in ambient air, foods, drinking water, soil, and dust. Segments of the general population at highest risk of health effects from Pb exposure are preschool-age children and pregnant women and their fetuses. Other segments of the general population with an increased exposure include individuals living near sites where Pb was produced or disposed. Some of the more important Pb exposures have occurred as a result of living in urban environments, particularly in areas near stationary emission sources (e.g., smelters); renovation of homes containing Pb-based paint; pica (the compulsive, habitual consumption of nonfood items); contact with interior Pb paint dust; occupational exposure; and secondary occupational exposure (e.g., families of workers in Pb industries). Higher exposures may also occur to residents living in close proximity to NPL sites that contain elevated levels of Pb.

The primary source of Pb in the environment has historically been anthropogenic emissions to the atmosphere. In 1984, combustion of leaded gasoline was responsible for approximately 90% of all anthropogenic Pb emissions. The United States gradually phased out the use of Pb alkyls in gasoline, and by 1990, auto emissions accounted for only 33% of the annual Pb emissions (EPA 1996b). Use of Pb additives in most motor fuels was totally banned after December 31, 1995 (EPA 1996a). The ban went into effect on February 2, 1996. The ban did not include off-road vehicles, including aircraft, racing cars, farm equipment, and marine engines. Pb additives are still used in fuels for piston driven airplane engines and it continues to be commercially available for other off-road uses. Atmospheric deposition is the largest source of Pb found in soils. Pb is transferred continuously between air, water, and soil by natural chemical and physical processes such as weathering, runoff, precipitation, dry deposition of dust, and stream/river flow; however, soil and sediments appear to be important sinks for Pb. Pb particles are removed from the atmosphere primarily by wet and dry deposition. The average residence time in the atmosphere is 10 days. Over this time, long-distance transport, up to thousands of kilometers, may take place. The speciation of Pb in these media varies widely depending upon such factors as temperature, pH, and the presence of humic materials. Pb is largely associated with suspended solids and sediments in aquatic systems, and it occurs in relatively immobile forms in soil.

5.2 PRODUCTION, IMPORT/EXPORT, USE, AND DISPOSAL

5.2.1 Production

The most important mineable Pb ore is galena (PbS), which is commonly associated with other minerals, typically zinc ores. Anglesite ($PbSO_4$) and cerussite ($PbCO_3$), formed by the weathering of galena, are two other important Pb minerals. Pb is processed from ore to refined metal in four steps: ore dressing; smelting; drossing; and refining. Ore dressing involves crushing, grinding, and beneficiation (concentration) (King et al. 2014).

Since 1998, U.S. production of Pb has shifted to the domestic secondary Pb industry (USGS 2014). Since 2014, primary Pb metal has not been produced in the United States (USGS 2016). The Doe Run Resources Corporation operated the last domestic primary Pb smelter-refinery facility in the United States at Herculaneum, Missouri and it was closed at the end of 2013. Pb-acid batteries are the dominant source of recoverable Pb scrap, accounting for nearly 100% of all secondary Pb (USGS 2016, 2019).

Domestic mines produced 368,000 metric tons of recoverable Pb in 2014, a more than 11% increase from 2013. Nearly all of the secondary Pb produced in 2014 was by 7 companies operating 12 plants in Alabama, California, Florida, Indiana, Minnesota, Missouri, New York, Pennsylvania, Tennessee, and Texas (USGS 2016). Secondary (recycled) Pb, derived from mainly scrapped Pb-acid batteries, accounted for all of the domestic refined Pb production in 2014. Due to plant closings, U.S. production of secondary refined Pb decreased in 2014 by 11% to 1.02 metric tons, from 1.5 metric tons in 2013 (USGS 2016).

World mine production of Pb was 4.91 million metric tons in 2014, a decrease of 9% from 2013. The United States accounted for approximately 8% of global mine production in 2014. The United States ranked third in global mine production behind China and Australia, which accounted for 49 and 15%, respectively. World production of refined Pb (primary and secondary) was 10.6 million metric tons in 2014. China produced about 45% of global refined Pb in 2014 with the United States as the second leading world producer of refined Pb, accounting for 10% (USGS 2016). In 2017 and 2018 worldwide mine production of Pb was reported as 4.58 and 4.40 million metric tons, respectively (USGS 2019). As in previous years, China was the dominant producer accounting for nearly half of the world production.

Manufacturers and importers of Pb metal and selected Pb compounds are listed in Table 5-1. These data are from EPA's Chemical Data Access Tool (now called Chemical Data Reporting [CDR]), which provides information on chemicals submitted to the EPA under the Toxic Substance Control Act that are manufactured or imported into the United States. Manufacturing volumes for more recent years are not available in the CDR as most manufacturers have withheld these data as confidential business information; however, as in previous years, the U.S. Geological Survey (USGS) reported total Pb mined in the United States in its Minerals Commodity summaries and these data for 2015–2018 are provided in Table 5-2. According to the USGS, five Pb mines located in the state of Missouri along with five mines in Alaska, Idaho, and the state of Washington accounted for all domestic Pb mine production (USGS 2019).

Table 5-1. U.S. Manufacturers of Lead Metal and Selected Lead Compounds

Company	Location	Domestic manufacturing (pounds/year)
Lead		
5n Plus Inc.	Fairfield, Connecticut	36,671
Colfin Specialty Steel Corp.	New Brighton, Pennsylvania	2,552
Compliance Administrators & Project Services Inc.	Bloomington, California	848,008
Concorde/Interspace Battery	West Covina, California	348,998
Doe Run Co.	Herculaneum, Missouri	280,000,000
East Penn Manufacturing Co. Inc.	Lyon Station, Pennsylvania	194,537,569
Exide Technologies	Bristol, Tennessee	150,000

Table 5-1. U.S. Manufacturers of Lead Metal and Selected Lead Compounds

Company	Location	Domestic manufacturing (pounds/year)
	Columbus, Georgia	4,200,000
	Forest City, Missouri	84,000,000
	Fort Smith, Arkansas	3,600,000
	Frisco, Texas	140,000,000
	Kansas City, Kansas	9,100,000
	Los Angeles, California	230,000,000
	Manchester, Iowa	16,000,000
	Muncie, Indiana	160,000,000
	Reading, Pennsylvania	130,000,000
	Salina, Kansas	990,000
Gopher Resource	Eagan, Minnesota	310,000,000
	Tampa, Florida	38,000,000
Horsehead Holding Corp.	Chicago, Illinois	2,444,492
	Palmerton, Pennsylvania	3,867,016
	Rockwood, Tennessee	1,872,054
	Snelling, South Carolina	2,012,236
Johnson Controls	Canby, Oregon	36,832,250
	Geneva, Illinois	47,025,828
	Holland, Ohio	82,721,150
	Kernersville, North Carolina	204,679,893
	Middletown, Delaware	86,732,852
	Tampa, Florida	3,069,380
	Yuma, Arizona	359,977,380
Johnson Controls Distribution Center	Saint Joseph, Missouri	2,550,177
	St. Joseph, Missouri	266,151,342
Renco Group Inc.	Boss, Missouri	310,000,000
Sanders Lead Co., Inc.	Troy, Alabama	471,954,520
Stemar Investments Inc.	Butler, Pennsylvania	40,506
Yuasa Battery Inc.	Laureldale, Pennsylvania	1,492,754
Lead(II) nitrate		
American Pacific Corp.	Cedar City, Utah	42,500
Lead(II) oxide		
C&D Technologies Inc.	Attica, Indiana	18,657,255
	Leola, Pennsylvania	1,348,311
	Milwaukee, Wisconsin	48,491,557
Crown Battery Manufacturing Co.	Fremont, Ohio	25,600,000

Table 5-1. U.S. Manufacturers of Lead Metal and Selected Lead Compounds

Company	Location	Domestic manufacturing (pounds/year)
Fiamm Energy LLC	Waynesboro, Georgia	4,700,000
Hammond Group Inc.	Hammond, Indiana	3,585,529
	Pottstown, Pennsylvania	8,287,521
Renco Group Inc.	Boss, Missouri	7,700,000
Steel Dust Recycling	Millport, Alabama	2,000,000
Superior Battery Manufacturing	Russell Springs, Kentucky	16,866,793
Trojan Battery Co.	Lithonia, Georgia	38,540,700
	Santa Fe Springs, California	35,241,500
Lead(II) styphnate		
Alliant Techsystems Inc.	Lewiston, Idaho	78,767
Alliant Techsystems Operations LLC	Independence, Missouri	43,489
Lead(II) sulfate		
Crown Battery Manufacturing Co.	Fremont, Ohio	768,000
East Penn Manufacturing Co., Inc.	Corydon, Iowa	17,006,710
	Lyon Station, Pennsylvania	220,436,420
Johnson Controls	Canby, Oregon	6,098,880
	Geneva, Illinois	11,340,306
	Holland, Ohio	10,714,048
	Middletown, Delaware	5,749,910
	Tampa, Florida	5,506,240
	Yuma, Arizona	86,756
Johnson Controls Distribution Center	Saint Joseph, Missouri	306,021
	St. Joseph, Missouri	29,577,055
Palos Verdes Bldg Corp.	Augusta, Georgia	6,904,629
Superior Battery Manufacturing	Russell Springs, Kentucky	22,905,105
Trojan Battery Co.	Lithonia, Georgia	58,127,100
	Santa Fe Springs, California	53,083,500
Lead(II) chloride		
Horsehead Holding Corp.	Monaca, Pennsylvania	1,891,700
	Palmerton, Pennsylvania	11,484,955

Source: EPA 2014d

Table 5-2. U.S. Lead Production 2015–2018

	Production volumes in metric tons			
	2015	2016	2017	2018
Mine, lead in concentrates	370,000	346,000	310,000	260,000
Secondary refinery, old scrap	1,050,000	986,000	1,130,000	1,300,000

Source: USGS 2019

Tables 5-3 (Pb) and 5-4 (Pb compounds) list the facilities in each state that manufacture or process Pb or Pb compounds, the intended use, and the range of maximum amounts of Pb that are stored on site. The data listed in Tables 5-3 and 5-4 are derived from the Toxics Release Inventory (TRI) (TRI18 2020). The data presented in Table 5-3 are for Pb metal and the data from Table 5-4 are for all Pb compounds. Facilities with ≥10 full-time employees in certain TRI-covered industry sectors (e.g., manufacturing) must submit data on releases and other waste management for TRI-listed chemicals (Pb and Pb compounds are TRI listed). Therefore, there are sources for Pb and Pb compounds not contained in the TRI database. In comparing TRI data with that of previous years, it is important to note that starting in 2001, the threshold for reporting Pb and all Pb compounds was reduced to 100 pounds, except for Pb contained in a stainless steel, brass, or bronze alloy. Previously, reporting was only required of facilities that manufactured or processed >25,000 pounds annually or that used >10,000 pounds annually. Beginning in 1998, additional industries were required to report, including metal mining, coal mining, electrical utilities, and Resource Conservation and Recovery Act (RCRA)/Solvent Recovery. Table 5-3 lists the producers of primary Pb metal and selected Pb compounds. Companies listed are those producing Pb compounds in commercial quantities >5,000 pounds or $10,000 in value annually. Table 5-4 shows the U.S. production volumes for Pb for 2010 through 2013. During this time, the primary Pb production declined, while secondary Pb production was relatively constant.

Table 5-3. Facilities that Produce, Process, or Use Lead

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AK	4	100	99,999	12
AL	105	0	999,999,999	1, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
AR	67	0	49,999,999	1, 2, 5, 6, 7, 8, 9, 11, 12, 13, 14
AZ	52	0	9,999,999	1, 3, 5, 7, 8, 9, 10, 11, 12, 14
CA	188	0	9,999,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
CO	33	0	99,999	1, 5, 6, 7, 8, 10, 11, 12, 14

Table 5-3. Facilities that Produce, Process, or Use Lead

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
CT	42	0	99,999	1, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
DC	6	0	9,999	1, 7, 8, 11, 12, 13, 14
DE	6	0	99,999	7, 12, 14
FL	223	0	999,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
GA	108	0	999,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
GU	1	0	99	1, 5
HI	2	0	99,999	8, 12
IA	114	0	9,999,999	1, 2, 5, 6, 7, 8, 9, 11, 12, 13, 14
ID	27	0	49,999,999	1, 2, 3, 5, 8, 9, 11, 12, 13, 14
IL	228	0	49,999,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
IN	161	0	999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
KS	70	0	9,999,999	1, 3, 5, 7, 8, 9, 10, 12, 13, 14
KY	79	0	999,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
LA	46	0	999,999	1, 2, 3, 5, 8, 9, 10, 11, 12, 13, 14
MA	55	0	999,999	1, 2, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MD	35	0	99,999	1, 5, 7, 8, 9, 10, 11, 12, 14
ME	20	0	99,999	1, 2, 3, 4, 5, 8, 9, 11, 12, 13, 14
MI	148	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
MN	123	0	99,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MO	92	0	9,999,999	1, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MS	62	0	999,999	1, 5, 7, 8, 9, 10, 12, 14
MT	15	0	99,999	1, 2, 5, 11, 12, 13, 14
NC	162	0	9,999,999	1, 2, 3, 5, 6, 7, 8, 9, 11, 12, 13, 14
ND	14	0	9,999	1, 5, 8, 9, 12, 14
NE	62	0	999,999	1, 2, 3, 5, 7, 8, 9, 11, 12, 13, 14
NH	30	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 14
NJ	43	0	99,999	1, 2, 5, 7, 8, 9, 11, 12, 13, 14
NM	17	0	999,999	1, 5, 6, 8, 9, 10, 11, 12, 14
NV	28	0	999,999	1, 2, 4, 5, 8, 9, 12, 13, 14
NY	140	0	999,999	1, 2, 3, 5, 7, 8, 9, 10, 11, 12, 13, 14
OH	237	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
OK	54	0	999,999	1, 2, 3, 5, 7, 8, 11, 12, 14
OR	48	0	9,999,999	1, 2, 3, 5, 7, 8, 10, 11, 12, 13, 14
PA	182	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
PR	5	100	9,999	2, 3, 8, 9, 12
RI	15	0	999,999	1, 5, 7, 8, 9, 10, 11, 12, 13, 14
SC	81	0	999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
SD	14	0	99,999	1, 2, 5, 7, 8, 9, 13, 14

Table 5-3. Facilities that Produce, Process, or Use Lead

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
TN	110	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
TX	326	0	49,999,999	1, 2, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
UT	50	0	999,999	1, 5, 6, 8, 9, 10, 11, 12, 13, 14
VA	98	0	999,999	1, 2, 3, 5, 7, 8, 9, 10, 11, 12, 13, 14
VI	3	0	99	7, 9, 14
VT	11	0	99,999	1, 2, 3, 5, 8, 11, 12, 13, 14
WA	57	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
WI	165	0	9,999,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
WV	31	0	999,999	1, 2, 5, 8, 9, 10, 11, 12, 13, 14
WY	12	0	99,999	1, 5, 6, 8, 9, 10, 11, 12, 13, 14

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

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

Source: TRI18 2020 (Data are from 2018)

Table 5-4. Facilities that Produce, Process, or Use Lead Compounds

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
AK	16	0	499,999,999	1, 5, 9, 12, 13, 14
AL	130	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
AR	68	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
AZ	65	0	49,999,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
CA	315	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
CO	76	0	49,999,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
CT	39	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
DC	1	100	999	14
DE	5	100	49,999,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
FL	150	0	49,999,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
GA	108	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
GU	3	0	99	1, 5, 7, 9, 12, 13, 14
HI	14	0	99,999	1, 2, 5, 7, 9, 12, 13, 14
IA	66	0	9,999,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14

Table 5-4. Facilities that Produce, Process, or Use Lead Compounds

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
ID	31	0	9,999,999	1, 2, 3, 4, 5, 7, 8, 11, 12, 13, 14
IL	166	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
IN	151	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
KS	40	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
KY	64	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
LA	85	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MA	53	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MD	35	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
ME	13	0	999,999	1, 2, 4, 5, 8, 9, 12, 13, 14
MI	102	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MN	50	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
MO	74	0	999,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
MP	1	0	99	1, 5, 12, 13, 14
MS	50	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 11, 12, 13, 14
MT	20	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 9, 12, 13, 14
NC	149	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
ND	20	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 12, 13, 14
NE	24	0	9,999,999	1, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14
NH	14	0	999,999	1, 2, 5, 7, 8, 9, 11, 12, 14
NJ	55	0	999,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
NM	18	0	49,999,999	1, 2, 3, 4, 5, 7, 8, 9, 11, 12, 13, 14
NV	51	0	49,999,999	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
NY	82	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
OH	175	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
OK	81	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
OR	55	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
PA	195	0	99,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
PR	11	0	99,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 12, 14
RI	19	0	99,999	1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
SC	101	0	49,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
SD	11	0	9,999,999	1, 3, 4, 5, 6, 7, 8, 9, 12, 13, 14
TN	89	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14
TX	330	0	499,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
UT	41	0	499,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
VA	84	0	49,999,999	1, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
VI	2	0	99	1, 5, 12, 0
VT	5	0	9,999	7, 8, 14
WA	76	0	9,999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14

Table 5-4. Facilities that Produce, Process, or Use Lead Compounds

State[a]	Number of facilities	Minimum amount on site in pounds[b]	Maximum amount on site in pounds[b]	Activities and uses[c]
WI	100	0	999,999	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14
WV	44	0	49,999,999	1, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14
WY	16	0	999,999	1, 2, 3, 4, 5, 8, 9, 12, 13, 14

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

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

Source: TRI18 2020 (Data are from 2018)

5.2.2 Import/Export

In 2014, 1,080 and 593,000 metric tons of Pb as base bullion and pigs and bars, respectively, were imported into the United States. Imports have increased since 2010 when 602 and 271,000 metric tons of Pb as base bullion and pigs and bars, respectively, were imported. In 2014, 65,100 metric tons, Pb content of Pb pigments and compounds were imported in the United States (USGS 2016). In 2015, 2016, 2017, and 2018 imports of Pb refined metal (unwrought) were reported as 521,000, 533,000, 658,000, and 580,000 metric tons, respectively (USGS 2019).

Exports of Pb in ore and concentrates and Pb materials, excluding scrap were 299,000 and 83,500 metric tons, respectively, in 2010 as compared to 365,000 and 61,300 metric tons, respectively, in 2014. In 2013 and 2014, 34,900 and 36,400 metric tons of Pb scrap were exported, respectively (USGS 2016). Total exports of Pb (Pb in concentrates and refined metal, unwrought gross weight) were reported as 406,000, 384,000, 293,000, and 324,000 metric tons in 2015, 2016, 2017, and 2018, respectively (USGS 2019).

5.2.3 Use

Pb may be used in the form of metal, either pure or alloyed with other metals, or as chemical compounds. The main uses of Pb and its compounds are in Pb-acid batteries, with most other applications using Pb alloys. The commercial importance of Pb is based on its physical properties, including its low melting point, ease of casting, high density, softness, malleability, low strength, ease of fabrication, acid resistance, electrochemical reaction with sulfuric acid, and chemical stability in air, water, and soil (King et al. 2014).

In the United States in 2014, Pb was consumed by over 70 companies to manufacture products such as ammunition; building-construction materials; covering for power and communication cable; Pb-acid storage batteries; Pb oxides for ceramics, chemicals, glass, and pigments; Pb sheet; and solder for construction, electronic components and accessories, metal containers, and motor vehicles (USGS 2016). In 2018, it was estimated that the Pb-acid battery industry accounted for >85% of the domestic consumption of Pb in the United States (USGS 2019). Pb-acid batteries were primarily used as starting-lighting-ignition (SLI) batteries for automobiles and trucks and as industrial-type batteries for standby power for computer and telecommunications networks and for motive power. Global consumption of refined Pb was 11.71 million metric tons in 2018, (USGS 2019).

Prior to the EPA beginning to regulate the Pb content in gasoline during the early 1970s, approximately 250,000 tons of organic Pb (e.g., tetraethyl Pb) were added to gasoline on an annual basis in the United States (Giddings 1973). These Pb-based “anti-knock” additives increased the octane rating of the gasoline and, as a result, increased engine efficiency (Giddings 1973). In 1971, the average Pb content for a gallon of gasoline purchased in the United States was 2.2 g/gallon (Giddings 1973). After determining that Pb additives would impair the performance of emission control systems installed on motor vehicles, and that Pb particle emission from motor vehicles presented a significant health risk to urban populations, EPA, in 1973, initiated a phase-down program designed to minimize the amount of Pb in gasoline over time. By 1988, the phase-down program had reduced the total Pb usage in gasoline to <1% of the amount of Pb used in the peak year of 1970 (EPA 1996a).

In 1990, a Congressional amendment to the Clean Air Act (CAA) banned the use of gasoline containing Pb or Pb additives as fuel in most motor vehicles. On February 2, 1996, the EPA incorporated the statutory ban in a direct final rule, which defined unleaded gasoline as gasoline containing trace amounts of Pb up to 0.05 g/gallon (EPA 1996a). The definition still allowed trace amounts of Pb, but expressly prohibited the use of any Pb additive in the production of unleaded gasoline. The term “lead additive” was defined to include pure Pb as well as Pb compounds (EPA 1996a). Although the regulatory action of Congress banned the use of leaded gasoline as fuel in motor vehicles, it did not restrict other potential uses of gasoline containing Pb or Pb additives (EPA 1996a). Gasoline produced with Pb additives continues to be made and marketed for use as fuels in aircraft, race cars, and non-road engines such as farm equipment engines and marine engines to the extent allowed by law (EPA 1996a), but tetraethyl Pb has not been produced in the United States since March 1991. All gasoline sold for motor vehicle use since January 1, 1996 has been unleaded (EPA 2020a).

Table 5-5 lists the uses of the specific Pb compounds identified in Chapter 4.

Table 5-5. Current and Former Uses of Selected Lead Compounds

Compound	Uses
Lead(II) acetate	Dyeing of textiles, waterproofing, varnishes, lead driers, chrome pigments, gold cyanidation process, insecticide, anti-fouling paints, analytical reagent, hair dye
Lead(II) azide	Primary detonating compound for high explosives, firing of Pb-based ammunition
Lead(II) bromide	Photopolymerization catalyst, inorganic filler in fire-retardant plastics, general purpose welding flux
Lead(II) carbonate	Polymerization catalyst, component of high pressure lubricating greases, coating on vinyl chloride polymers
Lead(II) chloride	Preparation of lead salts, lead chromate pigments, analytical reagent
Lead(II) chromate	Pigment in industrial paints, rubber, plastics, ceramic coatings; organic analysis
Lead(II) tetrafluoroborate	Salt for electroplating lead; can be mixed with stannous fluoborate to electroplate any composition of tin and lead as an alloy
Lead(II) iodide	Bronzing, printing, photography, cloud seeding
Lead molybdenum chromate	Analytical chemistry, pigments
Lead(II) nitrate	Lead salts, mordant in dyeing and printing calico, matches, mordant for staining mother of pearl, oxidizer in the dye industry, sensitizer in photography, explosives, tanning, process engraving, and lithography
Lead(II) oxide	Storage batteries, ceramic cements and fluxes, pottery and glazes, glass, chromium pigments, oil refining, varnishes, paints, enamels, assay of precious metal ores, manufacture of red lead, cement (with glycerol), acid-resisting compositions, match-head compositions, other lead compounds, rubber accelerator
Lead(II) phosphate	Stabilizing agent in plastics
Lead(II) styphnate	Primary explosive, firing of Pb-based ammunition
Lead(II) sulfate	Storage batteries, paints, ceramics, pigments, electrical and other vinyl compounds requiring high heat stability
Lead(II) sulfide	Ceramics, infrared radiation detector, semi-conductor, ceramic glaze, source of lead
Tetraethyl lead	Anti-knock agent in aviation gasoline

Sources: Boileau et al. 1987; Carr 1995; Carr et al. 2004; Davidson et al. 2014

Pb arsenate, basic Pb arsenate, and Pb arsenite were formerly used as herbicides, insecticides, or rodenticides. Until the 1960s, they were widely used to control pests in fruit orchards, especially apple orchards (EPA 2002c; PAN Pesticides Database 2004; Peryea 1998; Wisconsin DHS 2002). All insecticidal use of Pb arsenate was officially banned on August 1, 1988. However, all registrations for its insecticidal use had lapsed before that time.

5.2.4 Disposal

Secondary (recycled) Pb, derived mainly from scrapped Pb-acid batteries, accounted for 100% of refined Pb production in the United States in 2014. Almost all of the Pb recycled in 2014 was recovered by 7 companies operating 12 plants in Alabama, California, Florida, Indiana, Minnesota, Missouri, New York, Pennsylvania, Tennessee, and Texas (USGS 2016). More than 99% of all battery Pb is recycled and new batteries contain between 60 and 80% recycled Pb and plastic, respectively (BCI 2019). Scrap Pb is also recovered from dross, dust, residue, and sludge generated by smelting of metals, Pb pipe and sheet, printing materials, sheaths from power and telephone cable, and vehicle wheel weights (USGS 2014).

Disposal of wastes containing Pb or Pb compounds is controlled by several federal regulations (see Chapter 7). Pb is listed as a toxic substance under Section 313 of the Emergency Planning and Community Right to Know Act (EPCRA) under Title III of the Superfund Amendments and Reauthorization Act (SARA) (EPA 1988). Pb-containing waste products include storage batteries, ammunition waste, ordnance, sheet Pb, solder, pipes, traps, and other metal products; solid waste and tailings from Pb mining; items covered with Pb-based paint; and solid wastes created by mineral ore processing, iron and steel production, copper and zinc smelting, and the production and use of various Pb-containing products (EPA 1982a).

In the United States., federal laws require, used nickel cadmium (Ni-Cd) and lead (Pb) batteries to be managed as Universal Waste and recycled or disposed of in accordance under Title 40 Parts 266 and 273 of the Code of Federal Regulations (EPA 2020b). The Mercury-Containing and Rechargeable Battery Management Act (the Battery Act) of 1996 removed certain barriers to the recycling of batteries including small, sealed lead acid (SSLA) batteries (EPA 2002b). The intent was to provide the efficient and cost-effective collection and recycling or proper disposal of batteries to keep them out of the waste stream. The Act established uniform national labeling requirements, mandated that batteries under the Act be “easily removable” from consumer products where possible, made the Universal Waste Rule effective in all 50 states for the collection, storage, and transportation of batteries covered by the Battery Act, and required EPA to establish a public education program on battery recycling and the proper handling and disposal of used batteries (EPA 1997a).

According to data from the TRI, total disposal of Pb and Pb compounds varied during the period of 2005–2015 from 387 million pounds in 2009 to 832 million pounds in 2013, with an overall increase of 20% during this time period. The metal mining sector contributes most to the disposal of Pb and Pb compounds, with metal mines reporting 85% of total Pb and Pb compound releases in 2015.

5.3 RELEASES TO THE ENVIRONMENT

Facilities with ≥10 full-time employees in certain industry sectors (e.g., manufacturing) covered by the TRI (e.g., manufacturing) must submit data to TRI on releases and other waste management for TRI-listed chemicals (Pb and Pb compounds are TRI listed). Therefore, TRI data do not reflect all sources of Pb releases (EPA 2005a). TRI-covered facilities are required to report information to the TRI only if they employ the equivalent of ≥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 >25,000 pounds of any TRI chemical or otherwise uses >10,000 pounds of a TRI chemical in a calendar year (EPA 2005a).

Starting in 2001, the threshold to trigger reporting of Pb in most Pb compounds was reduced to 100 pounds. The higher threshold still applies to Pb contained in stainless steel, brass, or bronze alloys. The threshold for Pb is determined using the weight of the metal, whereas the threshold for Pb compounds is determined by the weight of the entire compound. Prior to 1998, only facilities classified within the SIC codes 20–39 (Manufacturing Industries) were required to report. After 1998, the industries required to report were enlarged to include other industrial sectors, such as metal mining, coal mining, electrical utilities, and hazardous waste treatment (EPA 2001).

Pb is a naturally-occurring element that is typically found combined in various minerals. It occurs in the Earth's crust primarily as the mineral galena (PbS), and to a lesser extent as anglesite ($PbSO_4$) and cerussite ($PbCO_3$) (Carr et al. 2004; Davidson et al. 2014; Haynes 2014). Pb minerals are found in association with zinc, copper, and iron sulfides as well as gold, silver, bismuth, and antimony minerals. It also occurs as a trace element in coal, oil, and wood. Typical Pb concentrations in some ores and fuels are: copper ores, 11,000 ppm; Pb and zinc ores, 24,000 ppm; gold ores, 6.60 ppm; bituminous coal, 3–111 ppm; crude oil, 0.31 ppm; No. 6 fuel oil, 1 ppm; and wood, 20 ppm (EPA 2001).

Leaded gasoline remains commercially available for off-road uses, including aircraft, racing cars, farm equipment, and marine engines. Currently, the largest contributor to atmospheric Pb emissions in the United States is piston-engine aircraft emissions (EPA 2016c). Industrial sources of Pb can result from the mining and smelting of Pb ores, as well as other ores in which Pb is a byproduct or contaminant. Fuel combustion also contributes to releases of Pb to the environment. As a result of these processes, Pb may be released to land, water, and air. Many of the anthropogenic sources of Pb have been eliminated or phased out because of Pb's persistence, bioaccumulative nature, and toxicity. These include Pb-based paint in 1978, Pb-containing pesticides in 1988, and Pb in gasoline for use in on-road vehicles in 1996. In early 2017, the use of Pb ammunition and Pb sinkers was banned on most federal lands; however, this ban was temporarily halted soon after. Because Pb does not degrade and remains in the environment long after its release, these former uses continue to be a potential source for Pb exposure.

5.3.1 Air

According to the TRI, in 2018, a total of 57,240 pounds of Pb were released to air from 4,064 reporting facilities (TRI18 2020). In addition, a total of 343,142 pounds of Pb compounds were released to air from 3,789 reporting facilities (TRI18 2020). Tables 5-6 and 5-7 list amounts of Pb and Pb compounds released from these facilities grouped by state, respectively.

Table 5-6. Releases to the Environment from Facilities that Produce, Process, or Use Lead[a]

		Reported amounts released in pounds per year[b]					Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
AK	4	1	0	0	22,766	0	22,167	600	22,767
AL	105	2,944	1,133	11	1,481,278	7,384	1,483,641	9,109	1,492,750
AR	66	2,318	107	0	176,475	730	177,674	1,955	179,629
AZ	52	103	129	0	42,201	331	40,794	1,970	42,765
CA	179	662	1,313	0	585,305	58,212	572,870	72,623	645,493
CO	34	296	20	0	108,291	51	108,414	244	108,658
CT	42	52	541	0	2	5,360	62	5,893	5,955
DC	6	1	2	0	6,493	1,000	6,464	1,031	7,495
DE	6	9	3	89	7,852	0	7,864	89	7,954
FL	221	693	89	1	98,375	15,249	84,880	29,527	114,408
GA	107	3,397	1,000	0	18,503	8,671	20,456	11,114	31,571
GU	1	28	0	0	0	0	0	0	28
HI	2	0	0	0	10,359	0	10,358	1	10,359
IA	114	10,241	80	849	6,205	12,559	14,396	15,538	29,934
ID	27	268	45	0	367,560	29,707	367,635	29,945	397,579
IL	228	3,147	2,286	27	649,364	43,310	613,450	84,683	698,133
IN	160	1,056	272	22	454,088	557,867	7,434	1,005,871	1,013,305
KS	70	536	26	0	24,749	729	13,856	12,183	26,039
KY	79	891	509	0	32,991	5,941	30,618	9,715	40,333
LA	45	511	967	13	22,715	462	22,763	1,905	24,669
MA	55	70	10,245	28	36,566	3,555	15,467	34,998	50,465
MD	34	25	59	0	13,986	86	13,467	689	14,156
ME	20	3	23	0	219	123	39	329	368
MI	147	2,095	396	11	41,090	5,359	19,114	29,837	48,951
MN	123	1,199	232	5	4,017	624	1,971	4,106	6,077
MO	92	483	225	1	20,478	729	12,844	9,072	21,915
MS	62	1,102	35	1	63,143	1,327	63,240	2,368	65,608
MT	13	16	2	0	36,873	0	36,872	20	36,892
NC	162	567	708	248	122,679	1,295	16,540	108,957	125,497
ND	13	1	229	0	6,038	7	6,130	144	6,275
NE	61	4,505	1,379	0	79,160	9,063	83,397	10,709	94,107
NH	29	17	6	0	726	26,933	743	26,939	27,682
NJ	42	198	27	17	3,286	4,976	1,419	7,086	8,505
NM	16	12	0	0	6,472	61	5,582	963	6,545
NV	27	502	1	1	2,554,095	150	2,554,538	210	2,554,748

Table 5-6. Releases to the Environment from Facilities that Produce, Process, or Use Lead[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
NY	140	499	89	0	21,001	9,770	14,560	16,799	31,359
OH	236	4,321	540	232	68,683	33,151	34,929	71,997	106,926
OK	54	451	11	14	81,644	3,422	81,961	3,581	85,542
OR	48	77	45	0	1,495,571	742	1,494,474	1,961	1,496,435
PA	183	6,456	1,908	425	167,238	486,904	12,164	650,767	662,931
PR	4	4	0	0	3,777	47	3,780	48	3,828
RI	14	10	1	0	3	5,086	10	5,091	5,101
SC	80	517	212	0	43,446	947	35,803	9,319	45,123
SD	14	100	0	0	0	25	100	25	126
TN	109	756	222	1	33,156	8,866	21,918	21,083	43,001
TX	325	1,223	1,027	2,833	468,642	45,651	468,737	50,638	519,376
UT	49	835	24	0	95,346	80	89,493	6,793	96,286
VA	93	1,176	574	17	184,110	26,197	83,245	128,828	212,074
VI	3	2	0	0	31	0	24	8	32
VT	11	1	6	0	16,304	397	16,289	419	16,708
WA	54	115	184	0	382,285	212,687	371,196	224,075	595,272
WI	163	1,921	638	0	35,520	13,924	5,242	46,762	52,004
WV	30	410	190	52	144,313	38	144,201	803	145,004
WY	10	416	1	0	47,579	1	47,733	263	47,996
Total	4,064	57,240	27,764	4,901	10,393,047	1,649,785	9,363,048	2,769,688	12,132,737

[a] The TRI data should be used with caution since only certain types of facilities are required to report. This is not an exhaustive list. Data are rounded to nearest whole number.
[b] Data in TRI are maximum amounts released by each facility.
[c] Post office state abbreviations are used.
[d] Number of reporting facilities.
[e] The sum of fugitive and point source releases are included in releases to air by a given facility.
[f] Surface water discharges, waste water treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).
[g] Class I wells, Class II-V wells, and underground injection.
[h] Resource Conservation and Recovery Act (RCRA) subtitle C landfills; other onsite landfills, land treatment, surface impoundments, other land disposal, other landfills.
[i] Storage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown
[j] The sum of all releases of the chemical to air, land, water, and underground injection wells.
[k] Total amount of chemical transferred off0site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI18 2020 (Data are from 2018)

Table 5-7. Releases to the Environment from Facilities that Produce, Process, or Use Lead Compounds[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
AK	15	19,345	139	0	499,679,935	1,006	499,696,960	3,465	499,700,425
AL	130	6,801	4,859	0	1,254,651	4,503	1,167,133	103,682	1,270,814
AR	67	5,268	1,098	0	481,839	53,191	239,131	302,264	541,396
AZ	64	64,290	103	0	27,547,626	7,110	27,600,302	18,827	27,619,128
CA	305	5,816	435	0	2,656,973	27,145	1,127,596	1,562,773	2,690,369
CO	76	2,931	244	1	10,195,005	239	10,180,490	17,930	10,198,420
CT	39	445	565	0	32,116	14,510	708	46,929	47,637
DC	1	0	0	0	24	0	24	0	24
DE	5	127	17	0	489	1,070	599	1,103	1,703
FL	150	5,429	1,691	1	972,024	2,470	946,498	35,116	981,615
GA	107	6,011	4,228	0	573,267	242	361,887	221,860	583,747
GU	3	30	0	0	1,510	0	1,540	0	1,540
HI	14	2,406	35	18	57,605	4	57,568	2,501	60,068
IA	66	3,101	568	4	98,007	47,899	35,369	114,209	149,578
ID	31	1,337	105	0	587,617	1,071	588,601	1,529	590,130
IL	163	10,458	3,316	158	1,044,749	382,696	219,957	1,221,420	1,441,377
IN	147	26,832	60,668	86	3,554,799	1,702,496	1,345,912	3,998,968	5,344,880
KS	40	2,171	126	35	33,471	7,290	31,594	11,499	43,092
KY	64	15,233	719	272	758,794	116,393	739,954	151,456	891,411
LA	82	8,193	8,065	303	694,825	3,582	504,890	210,078	714,968
MA	53	888	357	0	20,874	2,919	2,655	22,382	25,037
MD	35	209	203	0	4,611	55,261	1,443	58,842	60,285
ME	13	750	510	0	23,485	8,099	4,729	28,116	32,844
MI	100	4,220	979	13	2,091,026	18,914	1,514,558	600,594	2,115,152
MN	50	2,961	291	0	980,965	2,228	97,635	888,810	986,445
MO	73	12,990	1,919	206	18,739,390	243,396	18,118,610	879,290	18,997,900
MP	1	2	0	0	1	0	3	0	3
MS	50	2,260	1,770	197,869	261,610	3,066	235,034	231,540	466,575
MT	20	3,438	20	4,503	6,091,825	581	6,092,803	7,565	6,100,367
NC	145	10,843	973	0	728,520	35,993	551,282	225,047	776,329
ND	19	4,103	19	0	90,847	5,984	76,746	24,207	100,953
NE	24	1,473	125	0	50,473	529	46,977	5,623	52,600
NH	14	20	23	0	1,509	6,243	94	7,702	7,796
NJ	54	871	12,266	0	410,073	234,474	62,349	595,336	657,685
NM	17	902	283	1	2,978,296	31,276	2,893,375	117,383	3,010,758
NV	49	9,161	1	0	49,276,056	40	49,282,572	2,685	49,285,258
NY	82	2,294	1,167	0	209,644	26,579	132,597	107,087	239,684

Table 5-7. Releases to the Environment from Facilities that Produce, Process, or Use Lead Compounds[a]

		Reported amounts released in pounds per year[b]							
							Total release		
State[c]	RF[d]	Air[e]	Water[f]	UI[g]	Land[h]	Other[i]	On-site[j]	Off-site[k]	On- and off-site
OH	175	9,346	17,583	5,952	1,483,307	263,386	1,093,350	686,224	1,779,574
OK	80	11,075	378	58	307,485	945	307,785	12,157	319,941
OR	55	1,591	2,127	0	5,176	1,827	6,272	4,449	10,721
PA	195	13,623	15,874	0	3,488,454	229,953	3,176,873	571,031	3,747,904
PR	11	759	13	0	236	1,300	772	1,536	2,308
RI	19	23	15	0	1,018	124	24	1,155	1,179
SC	100	4,198	1,783	0	459,973	38,536	101,830	402,660	504,490
SD	11	149	0	0	1,757,845	9	1,757,955	48	1,758,003
TN	89	4,151	2,653	184	3,375,608	11,693	2,953,656	440,633	3,394,289
TX	323	14,625	2,732	2,309	1,365,901	3,582	1,170,161	218,988	1,389,149
UT	40	17,378	657	0	185,430,965	41,774	185,290,906	199,868	185,490,774
VA	82	10,922	8,300	0	201,670	12,410	206,084	27,217	233,301
VI	2	13	0	0	0	0	13	0	13
VT	5	10	0	0	117	0	10	117	127
WA	76	4,205	843	0	2,976,519	10,118	2,925,078	66,607	2,991,685
WI	98	3,448	1,598	0	279,692	27,606	74,486	237,858	312,344
WV	44	3,235	920	0	362,089	1,142	240,520	126,866	367,385
WY	16	783	2	0	79,050	284	67,659	12,461	80,119
Total	3,789	343,142	163,361	211,975	833,759,635	3,693,189	823,333,608	14,837,694	838,171,302

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

RF = reporting facilities; UI = underground injection

Source: TRI18 2020 (Data are from 2018)

The decrease in national Pb emissions between 1970 to 2011 is estimated to be 99.6% (220,000 tons), which is mostly attributed to the elimination of leaded gasoline for on-road vehicles. Since 2000, nonroad engines and metals industrial processing have accounted for most of the anthropogenic Pb emissions in the United States (EPA 2015). Based on data from the National Emissions Inventory (NEI 2014), the following sectors contribute the largest portions of total Pb emissions in the United States: mobile-aircraft (63%), industrial processes–not elsewhere classified (6.8%), industrial processes–ferrous metals (6.8%), fuel combustion–electric generation–coal (5.5%), and industrial processes–non-ferrous metals (4.1%) (EPA 2016c). Historical trends of Pb emissions in the United States are provided in Table 5-8 (EPA 2015).

Table 5-8. Historic Levels of Lead Emissions to the Atmosphere in the United States (in Thousand Metric Tons)

	1970	1975	1980	1985	1990	1995	1999	2002	2005	2008	2011
On-road vehicles	172	130.2	60.5	18.1	2.17	2.05	1	0	0	0	0
Metals industrial processing	24.22	9.923	3.03	2.1	0.5	0.49	0.96	0.4	0.3	0.16	0.14
Fuel combustion	10.62	10.35	4.3	0.52	0.42	0.02	0	0.39	0.14	0.12	0.09
Nonroad engines	9.737	6.13	4.2	0.92	0.78	0.54	0.55	0.45	0.66	0.56	0.49
Other sources	4.331	3.053	2.12	1.31	1.11	0.83	0.84	0.43	0.25	0.11	0.1

Source: EPA 2015

According to the data from the NEI, the largest portions of total Pb emissions are in the U.S. mobile-aircraft sector. Murphy et al. (2008) studied weekly patterns of metals and other aerosol components using data collected from 2000 to 2006 at Interagency Monitoring of Protected Visual Environments (IMPROVE) sites, and these data suggested that Pb concentrations were impacted by piston aircraft emissions.

As indicated in Table 5-6, by the early 2000s, transportation (i.e., automotive) emissions were no longer the dominant source of Pb emitted to the atmosphere. When such emissions were prevalent, >90% (mass basis) of automotive Pb emissions from leaded gasoline were in the form of inorganic particulate matter (e.g., Pb bromochloride [PbBrCl]) and <10% (mass basis) were in the form of organolead vapors (e.g., Pb alkyls). In 1984, the average Pb content of gasoline was 0.44 g Pb/gallon (EPA 1986b); however, as of January 1986, the allowable Pb content of leaded gasoline dropped to 0.1 g Pb/gallon (EPA 1985d). Between January and June of 1990, the actual average Pb concentration in leaded gasoline was 0.085 g Pb/gallon, indicating consumption of approximately 230,000 kg of Pb for the production of 2.74 billion gallons of leaded gasoline. In the early 1980s, EPA allowed up to 0.05 g of Pb in a gallon of unleaded gasoline (EPA 1982b).

According to data from TRI, on-site air releases of Pb and Pb compounds varied over the same period from 431,311 pounds in 2014 to 1,037,265 pounds in 2006, with an overall decrease of 40%. In 2018, 91,028 pounds of Pb and Pb compounds were released to air. The electric utility and primary metals industry sectors contributed to this overall decrease; both sectors have decreased air Pb and Pb compounds releases by approximately 70% from 2005 to 2015. The primary metal sector, which includes iron and steel manufacturers and smelting operations, contributes the greatest quantity of Pb and Pb compounds to air releases (EPA 2017a, 2017b).

While Pb levels in paints for interior use have been restricted since the 1950s, older houses and furniture may still be covered with leaded paint. Releases from Pb-based paints are frequently confined to the area in the immediate vicinity of painted surfaces, and deterioration or removal of the paint by sanding or sandblasting can result in high localized concentrations of Pb dust in both indoor and outdoor air.

The largest volume of organolead vapors released to the atmosphere results from industrial processes; prior to its phaseout and ban, leaded gasoline containing tetraethyl Pb as an anti-knock additive was also a major contributor. Tetraalkyl Pb vapors are photoreactive, and their presence in local atmospheres is transitory. Halogenated Pb compounds are formed during combustion by reaction of the tetraalkyl Pb compounds with halogenated Pb scavenger compounds. These halogenated Pb compounds ultimately give rise to Pb oxides and carbonates in the environment (EPA 1985b). Tetraalkyl Pb compounds once contributed 5–10% of the total particulate Pb present in the atmosphere. Organolead vapors were most likely to occur in occupational settings (e.g., gasoline transport and handling operations, gas stations, and parking garages) and high-traffic areas (Nielsen 1984).

5.3.2 Water

According to the TRI, in 2018, a total of 27,764 pounds of Pb were released to water from 4,064 reporting facilities (TRI18 2020). In addition, a total of 163,361 pounds of Pb compounds were released to water from 3,789 reporting facilities (TRI18 2020). Tables 5-6 and 5-7 list amounts of Pb and Pb compounds released from these facilities grouped by state, respectively.

The following industry sectors accounted for the majority of release of Pb to surface water in 2018: chemicals (14%); paper (12%); primary metals (10%); transportation equipment (5%); and fabricated metals (2%). The following industry sectors accounted for the majority of release of Pb compounds to surface water in 2018: paper (20%); electric utilities (8%); primary metals (8%); metal mining (3%) (TRI18 2020). The trends in discharges of Pb and Pb compounds to surface water from 2001 to 2018 are presented in Table 5-9.

Table 5-9. U.S. Surface Water Discharges of Lead and Lead Compounds (Pounds/Year)

Year	Lead	Lead compounds
2001	45,871	97,479
2002	20,694	92,366
2003	21,314	109,299
2004	14,564	107,386
2005	15,883	100,778
2006	22,985	86,772
2007	16,745	82,815
2008	11,404	153,681
2009	9,886	73,683
2010	7,263	72,556
2011	7,086	77,568
2012	7,307	60,656
2013	6,327	76,053
2014	8,836	79,344
2015	5,264	70,981
2016	9,507	119,566
2017	11,901	100,552
2018	9,943	105,473

Source: EPA 2017c; TRI18 2020

Data reported by Environment and Climate Change Canada (2016) show that other industries, which include the iron and steel industry, oil and gas industry, and cement and concrete industry, contributed 136.9 tonnes of the total Pb released to water in 2014. This release includes 134.1 tonnes of Pb that were released when a dam securing a tailings pond from the Mount Polley mine in central British Columbia breached on August 4, 2014, spilling mining waste into Polley Lake and surrounding waters. Waste, pulp, paper, and paperboard industry, and non-ferrous smelting and refining were the next largest contributors (Table 5-10). In 2013, Pb releases to water were similar for other industries and waste.

Table 5-10. Canada Surface Water Discharges of Lead and Lead Compounds (Tonnes)

Year	Other industries	Waste	Pulp, paper, and paperboard industry	Non-ferrous smelting and refining	Other sources
2003	4.38	15.49	2.55	1.74	0.18
2004	3.97	11.53	2.84	2.26	0.26
2005	6.11	9.47	3.29	1.82	0.58
2006	5	9.9	2.35	1.65	0.24
2007	3.63	6.42	2.37	1.64	0.19
2008	4.76	11.58	2.42	2.04	0.16
2009	3.39	8.49	2.25	2.13	0.19
2010	3.21	11.97	2.12	1.45	0.14
2011	3.65	8.97	2.91	1.5	0.16
2012	4.66	4.69	2.8	1.75	0.12
2013	4.17	4.66	2.42	1.48	0.13
2014	136.92	5.11	1.85	1.77	0.13

Source: Environment and Climate Change Canada (2016)

Urban runoff and atmospheric deposition are significant indirect sources of Pb found in the aquatic environment. Pb reaching surface waters is sorbed to suspended solids and sediments (EPA 1982a; EPA 2006, 2014c).

Pb is released into surface water from Pb shot and Pb sinkers. A study of a shooting range in Southwestern Virginia found that the dissolved Pb content of surface water ranged up to 473 ppb, with the highest concentrations closest to the backstop (Craig et al. 1999). Upstream from the site, the Pb concentration was 0.5 ppb. In 1991, the U.S. Fish and Wildlife Service banned the use of Pb shot when hunting waterfowl, such as geese or ducks, in order to avoid releasing Pb directly to surface water.

5.3.3 Soil

According to the TRI, in 2018, a total of 10,393,047 pounds of Pb were released to the land, both on-site and off-site, by 4,064 reporting facilities (TRI18 2020). Table 5-6 lists amounts of Pb released from these facilities grouped by state. In addition, a total of 833,759,635 pounds of Pb compounds were released to land, both on-site and off-site, by 3,789 reporting facilities (TRI18 2020). Table 5-7 lists amounts of Pb compounds released from these facilities grouped by state. In addition, 27,764 and 211,975 pounds of Pb and Pb compounds, respectively, were injected underground.

Pb-containing material from home and commercial use may be sent to municipal landfills. It is important to note that land is the ultimate repository for Pb, and Pb released to air and water ultimately is deposited in soil or sediment. For example, Pb released to the air from leaded gasoline or in stack gas from smelters and power plants will settle on soil, sediment, foliage, or other surfaces. The heaviest contamination occurs near the highway, in the case of leaded gasoline, or near the facility, in the case of a power plant or smelter. Road dust contributes to Pb in soil. Pb concentrations were higher in surface soils within 1,000 m of roadways (134 kg/ha) as compared to outside the 1,000-m region (38.7 kg/ha) (Yesilonis et al. 2008). Wheel weights can contribute to releases of Pb along roadways. Aucott and Caldarelli (2012) estimated that approximately 12 tons of Pb as wheel weights are deposited on New Jersey roadways; however, they estimated that only a small amount enters the environment as small particulate from grinding. Root (2000) also estimated a rate of Pb deposition in Albuquerque, New Mexico as 50–70 kg/km/year. However, use of Pb wheel weights are on the decline due to legislation, voluntary phase-out, and new wheel technology (Aucott and Caldarelli 2012).

5.3.4 Paint

Although the sale of residential Pb-based paint was banned in the United States in 1978, flaking paint, paint chips, and weathered powdered paint, which are most commonly associated with deteriorated housing stock in urban areas, remain major sources of Pb exposure for young children residing in these houses, particularly for children afflicted with pica (the compulsive, habitual consumption of nonfood items) (Bornschein et al. 1986; EPA 1986b). Pb concentrations of 1–5 mg/cm^2 have been found in chips of Pb-based paint (Billick and Gray 1978), suggesting that consumption of a single chip of paint would provide greater short-term exposure than any other source of Pb (EPA 1986b). An estimated 40–50% of occupied housing in the United States may contain Pb-based paint on exposed surfaces (Chisolm 1986).

In the late 1980s, the U.S. Department of Housing and Urban Development (HUD) conducted a national survey of Pb-based paint in housing. The EPA subsequently sponsored a comprehensive technical report on the HUD-sponsored survey to provide estimates of the extent of Pb-based paint in housing. In the EPA report, a home is considered to have Pb-based paint if the measured Pb concentration on any painted surface is $\geq$1.0 mg/cm^2. The EPA report estimates that 64 million (±7 million) homes, or 83% (±9%) of privately-owned housing units built before 1980, have Pb-based paint somewhere in the building. Approximately 12 million (±5 million) of these homes are occupied by families with children under the age of 7 years. Approximately 49 million (±7 million) privately owned homes have Pb-based paint in their interiors. By contrast, approximately 86% (±8%) of all pre-1980 public housing family units have Pb-based paint somewhere in the building (EPA 1995b).

Damaged Pb-based paint is associated with excessive dust Pb levels. Approximately 14 million homes (19% of pre-1980 housing) have >5 square feet of damaged Pb-based paint, and nearly half (47%) of those homes have excessive dust Pb levels (EPA 1995b).

In the Cincinnati prospective Pb study of public and private low- and moderate-income housing, the Pb concentration ranges were: painted interior walls, 0.1–35 mg/cm^2; interior home surface dust, 0.04–39 mg/m^2 and 72–16,200 μg/g; interior home dustfall, 0.0040–60 mg/m^2/30 days; exterior dust scrapings, 20–108,000 μg/g; and dust on children's hands, 1–191 μg. The Pb levels in older private deteriorating or dilapidated housing were higher than the levels in newer public and rehabilitated housing (Clark et al. 1985).

Releases from Pb-based paints are frequently confined to the area in the immediate vicinity of painted surfaces, and deterioration or removal of the paint can result in high localized concentrations of Pb in dust in air (from sanding and sandblasting) and on exposed surfaces. A study was conducted in New Orleans where power sanding is a common practice during repainting old houses; median, 90th percentile, and maximum Pb concentrations in 31 study houses were 35, 126, and 257 mg/g, respectively (Mielke et al. 2001). Pb concentrations in dust and soil samples from one study of a house where the paint chips contained about 90 mg Pb/g were very high. If the house had been sanded down to bare wood, 7.4 kg of Pb would have been released to the environment. Disturbance of older structures containing Pb-based paints is now a significant contributor to total Pb releases.

The authors of a report of findings from NHANES III, conducted in 1988–1991, commented that of the multiple sources of exposure, Pb-based paint is the principal high-dose source of Pb. Exposure occurs not only through the direct ingestion of flaking and chalking paint, but also through the inhalation of dust and soil contaminated with paint (Brody et al. 1994). According to a study by the New York State Department of Health, renovation and remodeling activities that disturb Pb-based paints in homes can produce significant amounts of Pb dust, which can be inhaled or ingested (CDC 1997a).

5.4 ENVIRONMENTAL FATE

The atmosphere is the main environmental transport media for Pb that is deposited onto surface water and soils (EPA 2006, 2014c). Upon release to the atmosphere, Pb particles are dispersed and ultimately removed from the atmosphere by wet or dry deposition. Pb deposition is typically greatest closer to Pb emission sources. An important factor in determining the atmospheric transport of Pb is particle size distribution. Large particles settle out of the atmosphere more rapidly and are deposited relatively close to emission sources and smaller particles may be transported much farther distances. After deposition, particles may be resuspended and redeposited. The cycling of Pb in aquatic environments is governed by chemical, biological, and mechanical processes. The exchange between sediment and surface water will be affected by pH, ionic strength, formation of organic complexes with Pb ions, and oxidation-reduction potential of the environment (EPA 2006, 2014c).

5.4.1 Transport and Partitioning

Transport and partitioning of Pb in the environment is an interplay of various processes (EPA 2014c). Global atmospheric deposition of Pb peaked in the 1970s and has declined since then; however, these deposits are still in the environment and can be transported and partitioned between environmental compartments. Past and current releases of Pb to the air result in the deposition of Pb on land and in surface water. While soil is a repository for Pb, it is not a passive repository, and resuspension of Pb contaminated soil-derived dust particulates can contribute to Pb exposure (Laidlaw and Filippelli 2008; Laidlaw et al. 2012). Pb in soil can be washed off surfaces into waters, and within water, it can partition between water and sediments (EPA 2006, 2014c).

Air. EPA (2006) summarized that the major pathway for the transport of Pb in the environment is the atmosphere and that airborne Pb tends to be in the form of submicron aerosols, which can travel large distances. After release to the atmosphere, Pb particles are dispersed and ultimately removed from the atmosphere by wet or dry deposition. Dry deposition was the major removal process for Pb in coarse particulate matter and wet deposition was the most important removal process for fine particulate matter. Soil-bound Pb and contaminated road dust can be resuspended and can be a significant source of airborne Pb in areas near major sources of Pb emissions (EPA 2006, 2014c).

In the atmosphere, non-organic compounds of Pb exist primarily in the particulate form. The median particle distribution for Pb emissions from smelters is 1.5 μm, with 86% of the particle sizes under 10 μm

(Corrin and Natusch 1977). The smallest Pb-containing particulate matter (<1 μm) is associated with high-temperature combustion processes. Upon release to the atmosphere, Pb particles are dispersed and ultimately removed from the atmosphere by wet or dry deposition. Approximately 40–70% of the deposition of Pb is by wet fallout; 20–60% of particulate Pb once emitted from automobiles is deposited near the source. An important factor in determining the atmospheric transport of Pb is particle size distribution. Large particles, particularly those with aerodynamic diameters of >2 μm, settle out of the atmosphere more rapidly and are deposited relatively close to emission sources (e.g., 25 m from the roadway for those size particles emitted in motor vehicle exhaust in the past); smaller particles may be transported thousands of kilometers away from the emission source.

The amount of Pb scavenged from the atmosphere by wet deposition varies widely; wet deposition can account for 40–70% of Pb deposition depending on such factors as geographic location and amount of emissions in the area (Nielsen 1984). An annual scavenging ratio (concentration in precipitation, mg/L, to concentration in air, $\mu g/m^3$) of 0.18×10^{-6} has been calculated for Pb, making it the lowest value among seven trace metals studied (iron, aluminum, manganese, copper, zinc, cadmium); this indicates that Pb (which initially exists as fine particles in the atmosphere) is removed from the atmosphere by wet deposition relatively inefficiently.

While Pb particles from automobile emissions are quite relatively small (<0.1 μm in diameter), they may coagulate to form larger particulates (Chamberlain et al. 1979). Pb has been found in sediment cores of lakes in Ontario and Quebec, Canada far from any point sources of Pb releases, suggesting that long-range atmospheric transport was occurring (Evans and Rigler 1985). Sabin and Schiff (2008) reported that median dry deposition fluxes along a coastal transect in southern California ranged from 0.52 to 14 $\mu g/m^2$-day in 2006. Pb fluxes ranged from 20 to 330 $\mu g/m^2$-day in 1975. Osterberg et al. (2008) reported elevated concentrations of Pb in a 1970–1998 ice core from the summit of Mt. Logan, Canada, and indicated that elevated levels correspond to increased industrial activity in Asia over the same time period. Mean Pb concentrations in the 1970–1998 portion were 68.9 ng/L, more than 10-fold above the natural background (5.6 ng/L).

Pb in soil in urban areas of older cities may be a source of airborne Pb (Laidlaw and Filippelli 2008). Studies of the Pb species found in airborne particulate matter collected in El Paso, Texas found that Pb-humate was the dominant form of Pb in air samples. Pb-humate, a stable, sorbed complex formed in the humus fraction of Pb contaminated soil, is the major Pb species in soils in El Paso (Pingitore et al. 2009). In a review, Cho et al. (2011) noted that, over the past 40 years, lead-bound air particulates have shifted to larger air particulate sizes as concentrations of Pb in urban areas have decreased. They note that this shift has occurred as the use of leaded gasoline was phased-out and that industrial emissions and resuspension of road dust became more important sources of Pb. In addition to soil-derived dust, re-entrainment of dusts near highways and deteriorating Pb-based paint from elevated steel structures can contribute to airborne Pb (Sabin et al. 2006; Weiss et al. 2006). Studies suggest that there is long-range transport of Pb bound to particulate matter from industrial emissions. Dust samples from surface glaciers and in dust traps in remote areas on the west coast of New Zealand's South Island were identified as being both Australian and New Zealand in origin. Samples were enriched in metals, including Pb, and the degree of metal enrichment indicted that they were transported from eastern Australia (Marx et al. 2008).

Water. The amount of soluble Pb in surface waters depends upon the pH and the ionic strength of the water. Equilibrium calculations show that at pH >5.4, the total solubility of Pb is approximately 30 μg/L in hard water and approximately 500 μg/L in soft water. Sulfate ions, if present in soft water, limit the Pb concentration in solution through the formation of Pb sulfate. Above pH 5.4, the Pb carbonates, $PbCO_3$ and $Pb_2(OH)_2CO_3$, limit the amount of soluble Pb. The carbonate concentration is in turn dependent upon the partial pressure of carbon dioxide, pH, and temperature (EPA 1986b).

A significant fraction of Pb carried by river water is expected to be in an undissolved form, which can consist of colloidal particles or larger undissolved particles of Pb carbonate, Pb oxide, Pb hydroxide, or other Pb compounds incorporated in other components of surface particulate matter from runoff. Pb may occur either as sorbed ions or surface coatings on sediment mineral particles, or it may be carried as a part of suspended living or nonliving organic matter in water.

Sediment and Soil. EPA (2006, 2014c) reviewed and summarized the factors affecting the behavior of Pb in soil. While Pb is relatively immobile in soil and has a long retention time in most soils, it has some capacity to leach through the soil column and potentially contaminate groundwater. Pb sorbs strongly to soil components and is only weakly soluble in pore water, making the leaching of Pb in soil a slower process as compared to other contaminants. Various soil conditions and characteristics affect the sorbing capacity of the soil and the solubility of contaminants including hydraulic conductivity of the soils, composition of the soil solution, organic matter, clay mineral content of the soil, pH, and microbial activity (EPA 2006). In soil, Pb can be partitioned between the soil water, precipitate forms, secondary iron and manganese oxides, carbonates, organic matter, sulfides, or the surfaces of clay, humus, or silicate particles. Pb adsorbed to the surfaces of colloid soil particles (e.g., organic matter, clay, oxides, and carbonates) are the most labile fraction. High chloride content in soil also enhances Pb solubility. At low pH, metal species bound to carbonates, hydroxides, and other soil components are more likely to dissolve into solution, increasing rates of Pb migration through the soil. EPA (2014c) reported that soil pH is the most important factor affecting solubility, mobility, and phytoavailability of Pb in soil; however, reducing conditions (e.g., anoxia) in soil also increase Pb mobility. In addition, dissolved organic matter is more important than iron oxyhydroxides in Pb mobility in soil.

The fate of Pb in soil is affected by the adsorption at mineral interfaces, precipitation of sparingly soluble solid forms of the compound, and formation of relatively stable organic-metal complexes or chelates with soil organic matter. These processes are dependent on such factors as soil pH, soil type, particle size, organic matter content of soil, presence of inorganic colloids and iron oxides, cation exchange capacity (CEC), and amount of Pb in soil (Getz et al. 1977; Reddy et al. 1995). Soil samples were extracted from the Powder River Basin in Wyoming to determine the relative distribution and speciation of Pb and other metals in acidic environments (Reddy et al. 1995). At near neutral pH, organic carbon-Pb complexes were the predominant species in the soil water extracts. At low pH, dissolved Pb in ionic form (Pb^{2+}) and ion pairs (e.g., $PbSO_4$) were the predominant species. It was concluded that the mobility of Pb will increase in environments having low pH due to the enhanced solubility of Pb under acidic conditions. The accumulation of Pb in most soils is primarily a function of the rate of deposition from the atmosphere. Most Pb is retained strongly in soil, and very little is transported through runoff to surface water or leached to groundwater except under acidic conditions (EPA 1986b; Getz et al. 1977). Clays, silts, iron and manganese oxides, and soil organic matter can bind metals electrostatically (cation exchange) as well as chemically (specific adsorption) (Reed et al. 1995). Although sorption to organic matter in soil limits the rate and extent of leaching, Pb may enter surface waters as a result of erosion of Pb-containing soil particulates. Pb bromochloride, the primary form of Pb emitted from motor vehicles, which once burned leaded gasoline in the presence of organohalogen scavenger compounds, is converted to the less-soluble Pb sulfate either by reactions in the atmosphere or by reactions at the soil surface, thus limiting its mobility in soil. It has been determined that Pb oxides, carbonates, oxycarbonates, sulfates, and oxysulfates become the most prominent constituents of aged automobile exhaust particles (i.e., those collected at locations more remote from traffic sources) (Ter Haar and Bayard 1971). Pb may also be immobilized by ion exchange with hydrous oxides or clays or by chelation with humic or fulvic acids in the soil (Olson and Skogerboe 1975). In soils with pH ≥5 and with at least 5% organic matter content, atmospheric Pb is retained in the upper 2–5 cm of undisturbed soil. Inorganic Pb may be bound into crystalline matrices of rocks and remain essentially immobile; it can also occur in water entrapped in soil macro- and micropores (Reed et al. 1995). In soil with high organic matter content and a pH of 6–8, Pb may form insoluble organic Pb complexes; if the soil has less organic matter at the same pH, hydrous Pb oxide complexes may form or Pb may precipitate out with carbonate or phosphate ions. At a pH of 4–6, the organic Pb complexes become soluble and leach out or may be taken up by plants (EPA 1986b). Entrainment or suspension of soil particles in moving air is another route of Pb transport (EPA 1982c). This process may be important in contributing to the atmospheric burden of Pb around some Pb smelting facilities and NPL sites that contain elevated levels of Pb in soil.

The downward movement of elemental Pb and inorganic Pb compounds from soil to groundwater by leaching is very slow under most natural conditions except for highly acidic situations (Getz et al. 1977). The conditions that induce leaching are the presence of Pb in soil at concentrations that either approach or exceed the CEC of the soil, the presence of materials in soil that are capable of forming soluble chelates with Pb, and a decrease in the pH of the leaching solution (e.g., acid rain) (Getz et al. 1977). Favorable conditions for leaching may be present in some soils near Pb smelting and NPL sites. Tetraalkyl Pb compounds, such as tetraethyl Pb, are insoluble in water and would not be expected to leach in soil. However, they can be transported through a soil column when it is present in a migrating plume of gasoline (USAF 1995). In aqueous media, tetraalkyl Pb compounds are first degraded to their respective ionic trialkyl Pb species and are eventually mineralized to inorganic Pb (Pb^{2+}) by biological and chemical degradation processes (Ou et al. 1995).

In a study of Pb migration in forest soils in Vermont, Miller and Friedland (1994) used Pb deposition time series and measurements of organic soil horizon Pb content made in 1966, 1980, and 1990 to compute dynamic response times for Pb storage in several types of soil. The authors concluded that maximum Pb concentrations in organic soil occurred around 1980, with concentrations of about 85 μg/g in soils of the northern hardwood forests of the study area and about 200 μg/g in soils of the spruce-fir forests. The large surge of atmospheric Pb deposited in these forests during the time when leaded gasoline was routinely used in motor vehicles is being redistributed in the soil profile rather than being retained in the organic horizon. Based on an analysis of Pb transit times through mineral soil horizons, the pulse of Pb may begin to be released to upland streams sometime in the middle of the next century (Miller and Friedland 1994). However, Wang et al. (1995) observed that Pb migration in forest soils is slowed considerably due to a decrease in solubility when Pb moves from the soil surface horizon to streams. Their results suggest that Pb is effectively trapped in the subsurface soil horizons, which may greatly reduce its release to streams.

Lewis et al. (2010) studied the distribution, chemical speciation, and mobility of Pb and antimony from small arms ammunition in a coarse-grained surface sand and reported that the transport of Pb was small in this soil type. Ninety-three percent of the mass of the bullets was found in the top 30 cm of the sand. Pb was mostly associated with the following grain sizes in decreasing order >5.0 mm (~3.3 g/kg), 1.2–5.0 mm (~1.5 g/kg), and <0.06 mm (~0.25 m/kg). In the 0.06–0.6 mm fractions, Pb concentrations were just above background levels (0.0004 g/kg). Declining concentrations with depth has also been observed in clay/loam shooting range soils (Vantelon et al. 2005). Pb in the fine fraction (<2 mm) shooting range soils also showed a depth distribution, with the highest concentrations in the top 10 cm (Cao et al. 2003a, 2003b; Hui et al. 2002; Lin et al. 1995; Perroy et al. 2014; Selonen et al. 2012). In a study of various contaminant levels in soil at a major training facility used for testing military tanks and munitions, Pb concentrations in the 0–15 cm soil depth ranged from 249.2 to 1,963.7 mg/kg (Berthelot et al. 2008).

Flooding events can change the spatial distribution of Pb in soil and sediments (EPA 2014c). Zahran et al. (2010) and Presley et al. (2010) reported variations in Pb concentrations in soil samples from schoolyards in New Orleans, Louisiana before and after Hurricanes Katrina and Rita in 2005, with some sites increasing and others decreasing in Pb concentrations. Forty-six census tracts in New Orleans were sampled before and after Hurricanes Katrina and Rita; 29 of these showed a decline in Pb concentrations, with 6 samples >400 mg/kg. Prior to these hurricanes, 15 of 46 samples had Pb concentrations >400 mg/kg. Across the tracts, the average median Pb concentration decreased from 328.5 to 203.33 mg/kg (Zahran et al. 2010). Presley et al. (2010) reported similar trends. Of the 17 schoolyard sites that were sampled, 7 sites had concentrations exceeding Pb concentrations of 400 mg/kg in June 2005, and in January 2006, Pb concentrations at 3 sites exceeded this concentration. The geometric mean concentration of the sites decreased from 290.0 to 207.4 mg/kg; however, at two sites, Pb concentrations increased from 804.0 to 1,740.0 mg/kg and from 1,090.0 to 2,500.0 mg/kg. During a 4-day storm event, 2,400 tonnes of suspended particulate matter were transported in a historical mining, ore processing, and smelting region in the Czech Republic that contained various metals including 2,954 kg of Pb (Žak et al. 2009).

Other Media. Plants and animals may bioconcentrate Pb, but biomagnification is not expected. In general, the highest Pb concentrations are found in aquatic and terrestrial organisms with habitats near Pb mining, smelting, and refining facilities; storage battery recycling plants; areas affected by high automobile and truck traffic; sewage sludge and spoil disposal areas; sites where dredging has occurred; areas of heavy hunting and fishing (Pb from spent shot or sinkers); and urban and industrialized areas. Pb may be present on plant surfaces as a result of atmospheric deposition; its presence in internal plant tissues indicates biological uptake from the soil and leaf surfaces. Although the bioavailability of Pb in soil to plants is limited because of the strong adsorption of Pb to soil organic matter, bioavailability increases with increased soil organic matter content and with decreased soil pH (more acidic). Plants grown in Pb-contaminated soils were shown to accumulate low levels of Pb in the edible portions of the plant from adherence of dusts and translocation into the tissues (Finster et al. 2004). Thirty-two different types of fruits or vegetables were grown in urban gardens with soils containing high Pb levels (27–4,580 mg/kg). Samples were harvested and washed with either water or detergents and analyzed for Pb content. Only one fruiting vegetable among 52 samples contained Pb levels greater than the detection limit of 10 μg/g in the edible portion. However, 39% of the leafy vegetables and herbs had Pb levels >10 μg/g in the edible shoot portion following washing of the vegetables with detergent and water (Finster et al. 2004).

Pb may be taken up in edible plants from the soil via the root system, by direct foliar uptake and translocation within the plant, and by surface deposition of particulate matter. The amount of Pb in soil that is bioavailable to a vegetable plant depends on factors such as cation exchange capacity, pH, amount of organic matter present, soil moisture content, and type of amendments added to the soil. Background agricultural soil Pb concentrations for major growing areas of the United States have been determined (Holmgren et al. 1993).

The influence of various combinations of soil amendments on Pb uptake by soybeans was studied for a metal-contaminated alluvial soil (Pierzynski and Schwab 1993). Addition of limestone was found to be most effective in reducing the bioavailability of metals (including Pb) as indicated by the reduction in labile soil metals, increased yields, and decreased soybean tissue metal content. Uptake of metals by lettuce and radishes grown in a loam soil spiked with cadmium chloride and Pb nitrate (from 100 to 1,000 mg/kg) was also studied (Nwosu et al. 1995). Results indicated that the mean uptake of Pb by lettuce increased as the concentration of Pb rose in the soil mixture. However, the uptake was low and this finding is inconsistent with other reports. Pb was not bioaccumulated by either plant regardless of soil Pb concentrations. The response of kidney bean growth to the concentration and chemical form of Pb in soils obtained near a zinc smelter in Japan has been studied (Xian 1989). It was found that the amount of Pb in the total plant (approximately 35–80 μg) correlated strongly with the concentration of Pb in the soil (0–240 mg/kg). The best relationship was found between the amount of metal uptake and the concentration of exchangeable and carbonate forms of Pb in the soil.

Uptake of Pb in animals may occur as a result of inhalation of contaminated ambient air or ingestion of contaminated plants. However, Pb is not biomagnified in aquatic or terrestrial food chains. Older organisms tend to contain the greatest body burdens of Pb. In aquatic organisms, Pb concentrations are usually highest in benthic organisms and algae, and lowest in upper trophic level predators (e.g., carnivorous fish). Exposure of a fresh water fish to several sublethal concentrations of Pb for a period of 30 days showed significant accumulation of Pb in the blood and tissues. The Pb accumulation in tissues was found to increase with Pb in water up to a concentration of 5 mg/L (μg/mL); at concentrations of 10 and 20 mg/L, the Pb accumulation in the tissues, although indicating an increase, was not proportional to the Pb concentration in water (Tulasi et al. 1992). High bioconcentration factors (BCFs) were determined in studies using oysters (6,600 for *Crassostrea virginica*), fresh water algae (92,000 for *Senenastrum capricornutum*), and rainbow trout (726 for *Salmo gairdneri*). However, most median BCF values for aquatic biota were significantly lower: 42 for fish, 536 for oysters, 500 for insects, 725 for algae, and 2,570 for mussels (Eisler 1988). Pb is toxic to all aquatic biota, and organisms higher up in the food chain may experience Pb poisoning as a result of eating Pb-contaminated food. Organolead compounds, such as trialkyl and tetraalkyl Pb compounds, are more toxic than inorganic forms and have been shown to bioconcentrate in aquatic organisms.

Biomagnification of organolead compounds has not been found to occur. Depuration is relatively rapid, with half-life values of 30–45 hours for rainbow trout exposed to tetramethyl Pb. Tetraalkyl Pb compounds are more toxic than trialkyl Pb compounds, and ethyl forms are more toxic than methyl forms (Eisler 1988). Isolation of a *Pseudomonas aeruginosa* strain designated CHL004, which is able to remove Pb from solidified media and soil, has been reported (Vesper et al. 1996). The rate of uptake of Pb nitrate by CHL004 was very rapid initially and then decreased greatly.

5.4.2 Transformation and Degradation

As an element, Pb cannot be degraded in the environment, but may undergo various precipitation or ligand exchange reactions. Pb will typically be found in compounds with oxygen and sulfur, and may undergo oxidation-reduction reactions under different environmental conditions. Under most environmental conditions, Pb will most likely exist in its Pb(II) oxidation state. Pb can be complexed by various ligands present in the environment (e.g., fulvic and humic acids). Despite forming complexes with organic matter, it is unlikely that it would be incorporated into organic compounds under environmental conditions. Transformations of Pb compounds that occur during their movement through the environment will be between various inorganic compounds.

Air. According to EPA (2014c), Pb accumulated on airborne mineral dusts can be transformed into different compounds during transport. It was also noted that Pb can accumulate on coarse particulate matter during transport in air and undergo chemical transformations. For example, Pb sulfate ($PbSO_4$), one of the main components of Pb-containing aerosols from coal combustion, can react with calcite ($CaCO_3$) in particulate matter to form various Pb carbonate compounds on the calcite surface. Another study included in the discussion noted that Pb levels in the PM_{10} fraction from dust storms collected in Israel were enriched with Pb at levels higher than those found in their source in the Sahara desert, suggesting that the dust samples accumulated Pb during transit between the Sahara desert and Israel (EPA 2014c).

Before the ban on sales of leaded gasoline, Pb particles were emitted to the atmosphere from automobile exhaust as Pb halides (mostly PbBrCl) and as double salts with ammonium halides (e.g., $2PbBrCl \cdot NH_4Cl$, $Pb_3[PO_4]_2$, and $PbSO_4$) (Biggins and Harrison 1979; Ter Haar and Bayard 1971). After 18 hours, approximately 75% of the bromine and 30–40% of the chlorine was released, and Pb carbonates, oxycarbonates, and oxides were produced. These Pb oxides are subject to further weathering to form additional carbonates and sulfates (Olson and Skogerboe 1975). Pb particles are emitted from mines and smelters primarily in the form of elemental Pb and Pb-sulfur compounds, $PbSO_4$, $PbO \cdot PbSO_4$, and PbS (Corrin and Natusch 1977; EPA 1986b; Spear et al. 1998). The Pb emitted from the combustion of waste oil was found to be in the form of $PbCl_2$, PbO, and elemental Pb (Pb^0) (Nerin et al. 1999). In the atmosphere, Pb exists primarily in the form of $PbSO_4$ and $PbCO_3$ (EPA 1986b).

While Pb is no longer added to gasoline for on-road use, the inorganic Pb degradation products of these organolead compounds may still be present in the environment. Based on the vapor pressure of tetraethyl Pb (0.26 mmHg at 25 °C) and tetramethyl Pb (26.0 mmHg at 20 °C), these two compounds exist almost entirely in the vapor phase in the atmosphere (Eisenreich et al. 1981). When exposed to sunlight, they decompose rapidly to trialkyl and dialkyl Pb compounds, and eventually to inorganic Pb oxides by a combination of direct photolysis, reaction with hydroxyl radicals, and reaction with ozone. The half-life of tetraethyl Pb in reactions with hydroxyl radicals during summer is approximately 5.7 hours, based on a rate constant of 6.8×10^{-11} cm^3/molecule-second (Nielsen et al. 1991). The half-life for tetramethyl Pb is about 65 hours, based on a rate constant of 5.9×10^{-12} cm^3/molecule-second. In the winter, both compounds have half-lives of up to several days since the concentration of atmospheric hydroxyl radicals is lower than in summer months (DeJonghe and Adams 1986).

Water. The fate of Pb in water will be determined by the conditions of the water, including acidity (pH), ionic strength, oxidation-reduction potential, flow rate, and amount and composition of suspended materials (EPA 2014c). The pH of water is an important factor in determining the fate of Pb in water. At neutral to more basic pH, Pb will tend to be complexed, precipitated, or sorbed to suspended sediments in water (EPA 2014c). Pb will form compounds of low solubility with the major anions found in natural waters. The maximum solubility of Pb in hard water is about 30 μg/L at pH>5.4 and the maximum solubility of Pb in soft water is approximately 500 μg/L at pH>5.4 (EPA 1977). In the environment, the divalent form (Pb^{2+}) is the stable ionic species of Pb. Hydroxide, carbonate, sulfide, and, more rarely, sulfate may act as solubility controls in precipitating Pb from water. At pH<5.4, the formation of Pb sulfate limits the concentration of soluble Pb in water, while at pH>5.4, the formation of Pb carbonates limits the amount of soluble Pb (EPA 1979). The relatively volatile organolead compound, tetramethyl Pb, may form as a result of biological alkylation of organic and inorganic Pb compounds by microorganisms in anaerobic lake sediments; however, if the water over the sediments is aerobic, volatilization of tetramethyl Pb from the sediments is not considered to be important because the tetramethyl Pb will be oxidized (EPA 1979).

The speciation of Pb was found to differ in fresh water and seawater. In fresh water, Pb may partially exist as the divalent cation (Pb^{2+}) at pHs below 7.5, but complexes with dissolved carbonate to form insoluble $PbCO_3$ under alkaline conditions (Long and Angino 1977). Even small amounts of carbonate ions formed in the dissolution of atmospheric CO_2 are sufficient to keep Pb concentrations in rivers at the 500 μg/L solubility limit (EPA 1979). Pb chloride and Pb carbonate are the primary compounds formed in seawater (Long and Angino 1977). The speciation of Pb in water is also dependent on the presence of other ligands in water. Pb is known to form strong complexes with humic acid and other organic matter (Denaix et al. 2001; Gao et al. 1999; Guibaud et al. 2003). Pb-organic matter complexes are stable to a pH of 3 with the affinity increasing with increasing pH, but decreasing with increased water hardness (EPA 1979). In seawater, there is the presence of Pb complexed to Fe-Mn oxides, which is due to the content of these oxides in seawater (Elbaz-Poulichet et al. 1984). Sorption of Pb to polar particulate matter in fresh water and estuarine environments is an important process for the removal of Pb from these surface waters. The adsorption of Pb to organic matter, clay, and mineral surfaces, and coprecipitation and/or sorption by hydrous iron and manganese oxides increases with increasing pH (EPA 1979).

Sediment and Soil. Pb in its naturally-occurring mineral forms is a component of many soils in the United States. The speciation of Pb in soils is dependent upon the properties of the soil. In a calcareous soil, $PbSO_4$ and $PbCO_3$ were shown to account for <5% of the total Pb content, whereas in roadside dust, $PbSO_4$, elemental Pb, Pb_3O_4, $PbO{\cdot}PbSO_4$, and $2PbCO_3{\cdot}Pb(OH)_2$ were present in significant quantities (Chaney et al. 1988). It was also reported that after adding 3,000–4,000 mg/kg of Pb in the form of

$PbSO_4$, subsequent extractions revealed that the Pb sulfate was rapidly transformed to other Pb compounds in the soil (Chaney et al. 1988).

Nearly all forms of Pb that are released to soil from anthropogenic sources, such as elemental Pb, $PbSO_4$, $PbCO_3$, PbS, $Pb(OH)_2$, $PbCrO_4$, and PbClBr, are transformed by chemical and biotic processes to adsorbed forms in soil (Chaney et al. 1988). The transformation process involves the formation of Pb complexes with binding sites on clay minerals, humic acid and other organic matter, and hydrous iron oxides (Chaney et al. 1988; Chuan et al. 1996; Sauve et al. 1997). The ability of soils to bind Pb is dependent on soil pH and the cation exchange capacity of the soil components (e.g., hydrous iron oxides on clay and organic matter) (Chaney et al. 1988; EPA 1986b). Only a small fraction (0.1–1%) of Pb appears to remain water-soluble in soil (Khan and Frankland 1983). The solubility of Pb in soil is dependent on pH, being sparingly soluble at pH 8 and becoming more soluble as the pH approaches 5 (Chuan et al. 1996). Between pH 5 and 3.3, large increases in Pb solubility in soil are observed. These changes in Pb solubility appear to correlate with the pH-dependent adsorption and dissolution of Fe-Mn oxyhydroxides. In addition to pH, other factors that influence Pb solubility in soil are total Pb content and the concentrations of phosphate and carbonate in soils (Bradley and Cox 1988; Ge et al. 2000; Pardo et al. 1990; Sauve et al. 1997).

Large particles of elemental Pb (e.g., shot and bullet fragments) degrade from weathering processes (Cao et al. 2003a, 2003b). Weathering includes physical transformation of larger particles to smaller particles (particle dissolution), as well as oxidation of the particle surface (coating) to PbO_2, with subsequent further oxidation to carbonates, phosphates, and sulfates (Cao et al. 2003a, 2003b; Hardison et al. 2004; Hashimoto 2013; Lewis et al. 2010; Lin et al. 1995; Rooney et al. 2007; Vantenlon et al. 2005). Particle dissolution rates for shotgun pellets in soils have been estimated to range from 1 to 20 mg/g pellet/year, depending on soil type, precipitation, and vegetation cover (Jorgenson and Willems 1987; Takamatsu et al. 2010).

Since the ban on the use of leaded gasoline, atmospheric Pb deposition to soil has decreased considerably. However, the deposited organolead compounds and their transformation products remain in the soil. Limited data indicate that tetraethyl and tetramethyl Pb are converted into water-soluble Pb compounds in soil through microbial metabolism (Ou et al. 1994). Using an Arredondo fine sand from Florida (92% sand, 7% silt, 1% clay, 11.8 g/kg organic carbon, pH 5.5), tetraethyl Pb was shown to degrade sequentially to monoionic triethyl Pb, diionic diethyl Pb, and eventually Pb^{+2} (Ou et al. 1994). Experiments were conducted using non-sterilized and autoclaved soil samples. The presence of monoionic triethyl Pb and diionic diethyl Pb was generally lower in the autoclaved samples, suggesting that both abiotic and biotic mechanisms are responsible for the degradation of tetraethyl Pb. At the end of a 28-day incubation period, no tetraethyl Pb was present in the soil; however, there were significant quantities of monoionic triethyl Pb and diionic diethyl Pb, which suggest that the degradation products are more persistent than the original species. Although tetraethyl and tetramethyl Pb are not expected to leach significantly through soil, their more water-soluble metabolites may be subject to leaching (EPA 1985a).

Pb content in plants is largely the result of atmospheric deposition. This is due to the strong retention of particulate matter on plant surfaces that is difficult to remove through washing (EPA 1977). Uptake of Pb into plant tissue appears to involve a combination of uptake from the leaf surface and uptake from roots, with the relative contribution of each pathway dependent on species and soil characteristics (Angelova et al. 2010; Bindler et al. 2008; Chrastny et al. 2010; Cui et al. 2007; Guyette et al. 1991; Hu and Ding 2009; Nwosu et al. 1995). Pb taken up by the root systems remains largely associated with root tissues (Comino et al. 2011; Businelli et al. 2011; Deng et al. 2004; Mellem et al. 2009; Murray et al. 2009; Nan and Cheng 2001; Sonmez et al. 2008; Wang et al. 2011). Translocation from roots to stem and leaf tissue has been shown to occur in some species (Peralta-Videa et al. 2009; Shaheen and Tsadilas 2009; Tamura et al. 2005; Wang et al. 2006; Zaprjanova et al. 2010). Eventually, the Pb will be returned to soil when these plants decay unless they are harvested (to possibly enter the food chain) or removed.

5.5 LEVELS IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to Pb depends, in part, on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of Pb in unpolluted atmospheres and in pristine surface waters are often so low as to be near the limits of current analytical methods. In reviewing data on Pb levels monitored or estimated in the environment, it should also be noted that the amount of chemical identified analytically is not necessarily equivalent to the amount that is bioavailable.

Table 5-11 shows the lowest limit of detections that are achieved by analytical analysis in environmental media. An overview summary of the range of concentrations detected in environmental media is presented in Table 5-12.

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

Media	Detection limit	Reference
Air	1.5 ng/cm^2 (XRF)	EPA, 1999, Method IO-3.3
	2.6 µg/sample	NIOSH 2017b, Method 7082
	6 µg/sample	NIOSH 1998, Method 7702
	0.02 µg/sample	NIOSH 1994c, Method 7105
	0.05 µg/sample	NIOSH 2016a, Method 7701
	0.062 µg/filter	NIOSH 2003c, Method 7300
	0.062 µg/filter	NIOSH 2003a, Method 7301
	0.023 µg/mL	NIOSH 2003b, Method 7303
	0.6 µg/sample	NIOSH 2014a, Method 7302
	1 µg/sample	NIOSH 2014b, Method 7304
	0.062 µg/sample	NIOSH 2015, Method 7306
	0.03 µg/mL	OSHA 2002, Method ID-121
	2.1 µg/sample	OSHA 2002, Method ID-125G
Drinking water	1.1 µg/L (ICP-AES) 0.02 µg/L (ICP-MS)	EPA 2003 Method 200.5 EPA 1994f Method 200.8
Surface water and groundwater	0.07 µg/L	EPA 1997b
	2.4 µg/L (GFAA) 0.28 µg/L (GFAA with preconcentration) 0.07 µg/L (ICP-MS)	EPA 1997b
	0.05 µg/L (ICP-MS) 60 µg/L (ICP-OES)	USGS 1989
	1 µg/L (GFAA) 1.1 µg/L (AVICP-AES)	USGS 1993
	10 µg/L (ICP) 100 µg/L (total recoverable, FLAA) 1 µg/L (whole water recoverable, GFAA) 0.5 µg/L (dissolved in water by GFAA) 100µg/L (suspended recoverable, FLAA) 100 µg/L (dissolved, FLAA)	USGS 1989
	0.6 µg/L (ICP-MS) 0.7 µg/L (GFAA) 10 µg/L (ICP-AES)	EPA 1994d
Soil/sediment	0.15 µg/g (ICP-MS) 0.2 µg/g (XRF) 0.2 µg/g (GF-AAS)	NOAA 1998
	10 µg/g (FLAA)	USGS 1989

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

Media	Detection limit	Reference
Wipes	0.042 µg/wipe	NIOSH 2003d, Method 9102
	0.02 µg/cm² for 100-cm² area (FLAA or ICP); 0.001 µg/cm² for 100-cm² area (GF-AAS)	NIOSH 1996a, Method 9100
	Range: 5–15 µg/wipe sample	NIOSH 2003e, Method 9105

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

AES = atomic emission spectroscopy; AVICP = axially viewed inductively coupled plasma; FLAA = flame atomic absorption; GFAA = graphite furnace atomic absorption; GF-AAS = graphite furnace-atomic absorption spectrometer; GRAV = gravimetry; ICP = inductively coupled plasma; MS = mass spectrometry; OES = optical emission spectrometry; Pb = lead; XRF = x-ray fluorescence

Table 5-12. Lead Levels in Water, Soil, and Air of National Priorities List (NPL) Sites

Medium	Median[a]	Geometric mean[a]	Geometric standard deviation[a]	Number of quantitative measurements	NPL sites
Water (ppb)	75	118	13.8	1,452	659
Soil (ppb)	1,110,000	885,000	19.7	1,453	661
Air (ppbv)	0.194	0.286	32.3	85	51

[a] 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

Four national monitoring networks collect data on Pb concentrations in ambient air to report to the Air Quality System (AQS). State and local agencies carry out monitoring at state and local monitoring stations (SLAMS). These data are primarily used to evaluate compliance with the National Ambient Air Quality Standard (NAAQS) for Pb. Pb levels are also monitored in the Chemical Speciation Network (CSN), Interagency Monitoring of Protected Visual Environments (IMPROVE), and National Air Toxics Trends Station (NATTS) networks. Pb concentrations in air are measured in three particulate matter (PM) size fractions: total suspended particles (TSP), PM_{10}, and $PM_{2.5}$. The CSN and IMPROVE networks monitor Pb in $PM_{2.5}$ and the NATTS network monitors Pb in PM_{10}. These networks are designed to meet different objectives than those of the Pb NAAQS monitoring network (EPA 2006, 2014c). EPA (2014c) analyzed data from these monitoring systems and presented data summaries for source-oriented (defined as near point sources and exceeded a defined emission threshold) and non-source-oriented Pb monitors across the United States for 2008–2010 (EPA 2014c). Maximum 3-month daily average Pb concentrations were calculated for non-source-oriented Pb-TSP monitors for 47 counties across the United States (1.5% of U.S. counties) and for source-oriented Pb-TSP monitors for 50 counties across the United States (1.6% of U.S. counties) during the period 2008–2010. Summaries of these analyses are presented in Table 5-13.

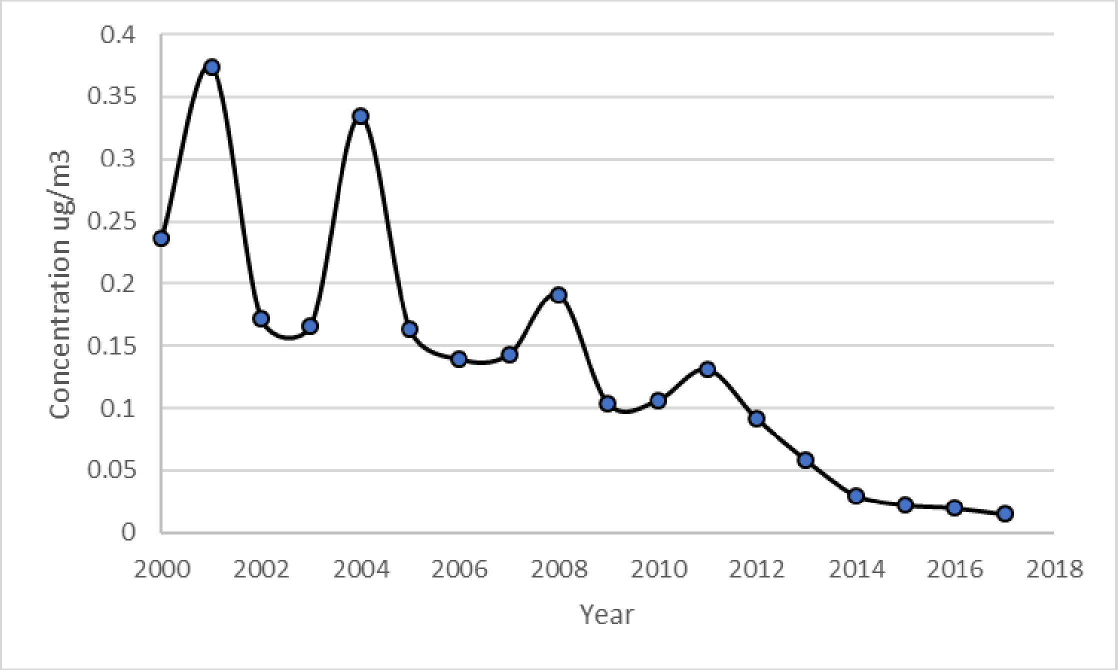
Table 5-13. Summary Data for Lead Monitors Across the United States, 2008–2010 (μg/m^3)

	Mean	Median	95th%	99th%	Maximum
Monthly (source-oriented)	0.20	0.063	0.86	1.6	4.4
Monthly (nonsource-oriented)	0.012	0.010	0.040	0.052	0.14

Source: EPA 2014c

Pb levels have been declining in the ambient air of the United States for several decades and according to the EPA, there has been approximately a 94% decrease since 2000 (EPA 2018a). Figure 5-3 shows the annual maximum 3-month average Pb level in the United States based upon data at 24 monitoring sites.

Figure 5-3. Annual Maximum 3-Month Average Representing the National Trend



Source: EPA 2018a

Data compiled from the EPA AQS database from 2015 to 2018 were used to calculate the percentile distribution of arithmetic mean 3-month averages at locations across the United States. These data are summarized in Table 5-14.

Table 5-14. Percentile Distribution of Mean Lead (TSP) Concentrations (μg/m^3) Measured in Ambient Air at Locations Across the United States

	Percentile				
Year	25th	50th	75th	95th	Maximum
2015	0.0036	0.0090	0.0216	0.0753	0.1942
2016	0.0038	0.0093	0.0220	0.0782	0.1466
2017	0.0039	0.0080	0.0190	0.0756	0.2087
2018	0.0035	0.0090	0.0313	0.1248	0.5574

TSP = total suspended particles

Source: EPA 2018b

Pb in indoor air is related to Pb in housedust, and predominant sources are outdoor air and degraded Pb-based paint (EPA 2006). Smoking can also contribute to higher concentrations of Pb in indoor air. Pb concentrations in air and dust in the indoor environment were measured in residential homes as part of the National Human Exposure Assessment Survey (NHEXAS) in EPA Region V (Indiana, Illinois, Michigan, Minnesota, Ohio, and Wisconsin). Mean (±1 SD) and median concentrations of Pb in indoor air from 213 residences were 15.2 ng/m^3 (37.6 ng/m^3) and 6.17 ng/m^3, respectively, with a maximum value of 293.5 ng/m^3 (Bonanno et al. 2001). The median Pb concentration in outdoor air was 8.84 ng/m^3 (Clayton et al. 2002). Pb concentrations were higher in households where one or more residents smoked indoors (mean concentration of 21.8 ng/m^3) as compared to households with nonsmoking residents (mean concentration of 7.79 ng/m^3) (Bonanno et al. 2001). In dust collected from the living areas of 238 residences, the mean (±1 SD) and median Pb concentrations were 467.4 μg/g (2,100 μg/g) and 131.6 μg/g, respectively, with a maximum value of 30,578 μg/g. Dust samples collected from window sills had mean (±1 SD) and median Pb concentrations of 987 μg/g (2,723 μg/g) and 207.5 μg/g, respectively, with a maximum value of 21,120 μg/g. For both indoor air and dust measurements, higher concentrations of Pb were correlated with dilapidated and suburban homes. Dixon et al. (2009) analyzed children’s exposures to residential dust Pb using data from the NHANES survey and associated demographics as well as smoking status to exposure levels. Children who resided in homes in which smoking occurred indoors had significantly ($p=0.015$) higher PbB levels than children who lived in homes of nonsmokers.

In another analysis of the NHEXAS EPA Region V data, Pellizzari et al. (1999) looked at potential differences in Pb concentrations in indoor air and personal air exposures between minorities (e.g., Hispanics and African-Americans) and nonminorities (e.g., Caucasian). Some differences were noted in the mean (±1 SD) Pb concentrations between minorities of 57 ng/m^3 (±24 ng/m^3) and nonminorities of 22 ng/m^3 (±3.4 ng/m^3) in personal air exposures, although the differences were not significant (p=0.147). Similarly, differences were noted between minorities (26±12 ng/m^3) and nonminorities (13±2.6 ng/m^3) in indoor air, although these were also not significantly different (p=0.266). When the age of the home was considered in the analysis, it was found that Pb concentrations were significantly (p=0.036) higher in homes built before 1940 than in homes built between 1960 and 1979, with mean (±1 SD) values of 46 ng/m^3 (±1.6 ng/m^3) and 13 ng/m^3 (±2.1 ng/m^3), respectively. The Pb concentrations measured in indoor air in homes built before 1940 were not significantly different from mean (±1 SD) Pb concentrations of 22 ng/m^3 (±5.1 ng/m^3) and 23 ng/m^3 (±5.1 ng/m^3) measured in indoor air in homes built between 1940 and 1959 and between 1980 and 1995, respectively.

5.5.2 Water

Pb has been monitored in surface water, groundwater, and drinking water throughout the United States and other countries. The concentration of Pb in surface water is highly variable depending upon sources of pollution, Pb content of sediments, and characteristics of the system (pH, temperature, etc.). Pb concentrations in surface water are generally higher in urban areas than in rural areas (EPA 1982c), and Pb measured in natural or “pristine” surface waters may be due to anthropogenic input. Western Airborne Contaminants Assessment Project (WACAP) data collected at five U.S. National Parks showed median Pb levels in surface waters ranging from 0.006 to 0.075 μg/L (EPA 2014c). The median Pb level in natural river water was 5 μg/L, with a range of 0.6–120 μg/L; however, lower Pb levels are to be expected after leaded gasoline was banned in 1985, which resulted in decreased rates of atmospheric deposition (Bowen et al. 1966; King et al. 2014). The National Academies of Science reported Pb concentration levels in surface water and groundwater (EPA 1986b). The mean Pb concentration level in surface water was 4 μg/L with a range from below the detection limit to 890 μg/L (EPA 2014c); concentrations >100 μg/L were observed near sources of urban runoff or industrial discharge. Mean levels of Pb in surface water measured at 50,000 surface water stations throughout the United States were 3.9 μg/L (based on 39,490 occurrences) (Eckel and Jacob 1988). Using the EPA Storage and Retrieval (STORET) database, from January 1, 2005 to May 16, 2005, Pb had been detected in surface water in Washington, Utah at concentrations of 20.5 and 142 μg/L and surface water from Salt Lake City, Utah at 7.75 μg/L (EPA 2005b). Pb was not detected above the detection limits in 224 other surface water samples obtained from various locations in Utah and Iowa over the sampling period (EPA 2005b). Pb content in groundwater is driven largely by the surrounding bedrock geochemistry; Pb concentrations are generally low in groundwater and natural springs ranging from below the detection limit to 100 µg/L (EPA 2014c). A USGS study of groundwater in the United States from 2000 to 2016 concluded that <1% of measured Pb concentrations are >15 µg/L, but when high levels are detected, they are typically associated with geographic locations where the Pb solubility potentials (the amount of Pb that could dissolve before a Pb mineral precipitates out of solution) are naturally high (Jurgens et al. 2019). Pb levels in seawater are typically in the range of 0.001–0.036 µg/L in the open ocean and about 0.050–0.30 µg/L in coastal waters influenced by anthropogenic activity (Angel et al. 2016).

Urban storm water runoff is an important source of Pb entering receiving waterways. Sources of Pb in runoff can be contributed to substantial direct atmospheric deposition, as well as indirect release from building materials, soil, and road dust, and industrial discharge. Pb is found in building material (brick, concrete, painted and unpainted wood, roofing, and vinyl), and automotive sources (brakes, used oil), which contribute to runoff (Davis et al. 2001). The largest contributing sources were siding and roofing. Soto-Jiménez and Flegal (2009) evaluated the sources of Pb pollution in the Gulf of California, northwest Mexico by sampling urban and rural areas for Pb levels and isotope ratios. Urban street dust (157 µg/g), agricultural soils (29.0 µg/g), and surface estuary sediments (35.6 µg/g) were all higher than natural bedrock (16.0 µg/g). Isotopic ratios in rural and soil runoff samples were comparable to natural Pb containing bedrock. Pb concentrations in the suspended particulate matter were measured in sewage effluent (132 µg/g), agricultural effluent (29.3 µg/g), river runoff (7.3 µg/g), and estuary water (68.3 µg/g). Urban, street dust, and sewage showed contributions from automotive emissions from past leaded gasoline combustion.

Pb in drinking water can derive from source water contamination as described above, but the more common source of Pb in drinking water is from internal corrosion of water distribution system piping and plumbing. Internal corrosion of Pb service lines, Pb-based pipe solder, brass meters and plumbing fixtures, and dissolution of existing protective scales contribute directly to Pb levels in drinking water. The Lead and Copper Rule (LCR) was promulgated in 1991 with the purpose of protecting public health by minimizing Pb and copper levels in drinking water, primarily by reducing water corrosivity (EPA 2004). The LCR established a Pb action level (AL) of 15 µg/L and a maximum contaminant level goal (MCLG) of zero. The Pb action level is based on feasibility of public water systems to control corrosion in their distribution systems and is not a health benchmark for Pb in drinking water. The Pb action level is exceeded if the concentration of the 90th percentile first draw tap sample (collected after a minimum stagnation period of 6 hours from high risk sites) exceeds 15 μg/L (EPA 2016a). If the Pb AL is exceeded, the LCR can require public water systems to take steps to minimize the risk of Pb exposure that may include source water monitoring/treatment, public education, water quality monitoring, implementing corrosion control treatment, and Pb service line replacement. In October 2019, EPA proposed significant changes to the LCR (EPA 2019a). These changes include: (1) identify areas most in need of remediation of Pb service lines; (2) establish a trigger level of 10 μg/L for requiring corrosion control in drinking water systems that do not currently treat for corrosion; (3) require water system to replace Pb service lines; (4) increase sampling reliability by prohibiting pre-stagnation flushing and other methods; (5) require systems to notify customers of action level exceedance within 24 hours; and (6) protect children in schools by expanding testing at drinking water outlets.

Analyses done in support of the short-term revisions to the LCR at the beginning of the 21st century suggest that in 2003, <2% of public water systems serving >3,300 people exceeded the Pb action level of 15 μg/L (EPA 2007a). Additionally, a 2004 study conducted by the EPA on LCR compliance monitoring for public water systems serving >3,300 people indicated that <4% of those systems exceeded the Pb action level (Hill 2011). It is important to note that states were not required until 2002 to report 90th percentile Pb concentrations to the EPA unless those samples exceeded the Pb AL; therefore, it is difficult to accurately compare differences between tap water Pb levels prior to LCR implementation and immediately following LCR implementation with current nationwide Pb concentration levels (Hill 2011). Nevertheless, the EPA evaluated water sample data from 166 large public water systems (systems serving >50,000 people) that exceeded the Pb AL in 1992 and 1993 (Hill 2011). Of the large systems that exceeded the Pb AL in 1992–1993, only 15 of those systems continued to exceed the Pb AL between 2000 and 2004, and their associated average 90th percentile Pb concentration levels significantly decreased from 32 to 8.2 μg/L.

The amount of Pb contained in pipes and plumbing fittings has been strictly regulated since 1986. Section 1417 of the Safe Drinking Water Act (SDWA) was amended to ban the use of service lines, pipe fittings, pipe solder, and fixtures that are not “Pb free” (not more than 0.2% Pb for pipe solder and flux, and not more than 8% Pb for pipe fittings and service lines) and are connected to a public water system and intended to provide water for human consumption. The 1996 Amendment broadened this ban by limiting the amount of leaching of Pb from new plumbing, and an industry standard was established. In 2011, the Reduction of Lead in Drinking Water Act amended Section 1417, revising the existing SDWA definition of “Pb free” and getting rid of the leaching certification requirement. Implemented in 2014, the act reduced the allowable level of Pb by “not more than a weighted average of 0.25 percent Pb when used with respect to the wetted surfaces of pipes, pipe fittings, plumbing fittings and fixtures.”

According to EPA's National Public Water Systems Compliance Report for calendar year 2013 (EPA 2013), 73% of public water systems in the United States, serving approximately 77% of the population, had no significant reported violations of any type. Significant violations include all violations of health-based standards, including violations of the maximum contaminant levels, treatment technique requirements, and significant monitoring and reporting requirements. In 2013, 7% of public water systems had no reported violations of health-based standards, and 5% of all health-based standard violations were LCR violations.

In the spring of 2014, the source of drinking water in the city of Flint, Michigan was switched from treated water obtained from Lake Huron to the Flint River. However, the treated water from the Flint River was more corrosive and did not contain corrosion inhibitors, which resulted in Pb leaching from the city’s aging service lines. Sampling data conducted in August of 2015 showed that the 90th percentile concentration of Pb in first-draw drinking water was 26.8 μg/L for 268 samples of tap water, which far exceeded the EPA AL of 15 μg/L (Pieper et al. 2018). In response to the high Pb levels in Flint drinking water, the city reconnected to the DWSD in October of 2015. By August of 2017, the 90th percentile concentration of Pb in first-draw tap water was 7.9 μg/L (Pieper et al. 2018).

5.5.3 Sediment and Soil

Pb is a naturally occurring metal found in the earth's crust at about 15–20 mg/kg (Goyer 2001). However, the concentration of Pb in the top layers of soil varies widely due to deposition and accumulation of atmospheric particulates from anthropogenic sources. The concentration of soil Pb generally decreases as distance from contaminating sources increases. The estimated Pb levels in the upper layer of soil beside roadways are typically 30–2,000 μg/g higher than natural levels, although these levels drop exponentially up to 25 m from the roadway (EPA 1986b). Soil adjacent to a smelter in Missouri had Pb levels in excess of 60,000 μg/g (Palmer and Kucera 1980). Soils adjacent to houses with exterior Pb-based paints have reported Pb levels >10,000 μg/g (EPA 1986b). As a result of Pb reactions with the soil, extractable Pb in surface soil samples (0–5 cm depth) from an agricultural area near a car battery manufacturing plant (taken at 0.3 km from the source) decreased from 117 to 1 μg/g within 1 year after the plant stopped operating (Schalscha et al. 1987). Soil collected by scraping the top 2.5 cm of soil surface near homes and streetside in Louisiana and Minnesota contained median Pb concentrations of >840 μg/g in New

Orleans and 265 μg/g in Minneapolis. In contrast, the small towns of Natchitoches, Louisiana, and Rochester, Minnesota had soil Pb concentrations of <50 and 58 μg/g, respectively. These data suggest that Pb-contaminated soil is a major source of Pb exposure in urban areas (Mielke 1993). As would be expected, soils in elementary school properties were also found to have the same pattern of Pb levels as the soils in the surrounding residences. Pb concentrations in soils collected from inner-city schools in New Orleans were higher (median concentration of 96.5 μg/g) than soils collected from mid-city (30.0 μg/g) and outer-city (16.4 μg/g) elementary schools (Higgs et al. 1999).

The former use of Pb in paints, particularly in older structures, is also a source of Pb in soil and within homes. Mielke and Gonzales (2008) reported median Pb concentrations of 76,603 mg/kg (464–317,151 mg/kg) and 416 mg/kg (24–63,313 mg/kg) for exterior and interior paints, respectively, in 40 paint chip samples collected from homes in metropolitan New Orleans. The authors noted that the age of the house is often used as a surrogate for the amount of Pb in paints; the mid-1920s being the peak use of leaded paint with declines until 1978. Demolition and renovation of buildings where leaded paint was used can result in transport of Pb to soil surrounding the building as well as indoor dust that contains Pb.

Pb concentrations were measured in residential transects through Lubbock, Texas. Pb concentrations through the city showed a trend of decreasing Pb concentrations with increasing distance from the city center, which also paralleled a decrease in the property age. The highest Pb concentrations in the city center were 90.0–174.0 mg/kg, with a median of 35.4 mg/kg, and decreased out to the farther part of the residential transect to 6.0–9.0 mg/kg. The highest concentrations outside city development were 4.9 mg/kg (Brown et al. 2008).

Studies conducted in Maryland and Minnesota indicate that within large, light-industrial, urban settings such as Baltimore, the highest soil Pb levels generally occur near inner-city areas, especially where high traffic flows have long prevailed (Mielke et al. 1983, 1984, 1989) and that the amount of Pb in the soil is correlated with the size of the city (Mielke 1991). In 1981, soil Pb levels in the Minneapolis/St. Paul inner-city area were 60 times higher (423 μg/g) than levels found in rural Minnesota (6.7 μg/g), with almost all the increase (95%) resulting from the combustion of leaded gasoline. A study conducted in Minneapolis, Minnesota, after the Pb content of gasoline had been significantly reduced, found that median soil Pb levels taken from the foundations of homes, in yards, and adjacent to the street were 700, 210, and 160 μg/g, respectively; median soil Pb concentrations in comparable samples from the smaller city of Rochester, Minnesota, did not exceed 100 μg/g at any location tested (Mielke et al. 1989). The Minneapolis data suggested that average Pb levels were elevated in soil samples taken from the foundations of homes, but that Pb levels were low (<50 μg/g) in areas where children could be expected to play, such as parks that were located away from traffic, but were higher in play areas around private residences. Soil samples taken from around the foundations of homes with painted exteriors had the highest Pb levels (mean concentrations of 522 μg/g), but levels around homes composed of brick or stucco were significantly lower (mean concentration 158 μg/g) (Schmitt et al. 1988). Severely contaminated soils (levels as high as 20,136 μg/g) were located near house foundations adjacent to private dwellings with exterior Pb-based paint. Elevated soil Pb concentrations were found in larger urban areas, with 27, 26, 32, and 42% of the soil samples exceeding 300 μg/g Pb in Duluth, inner-city North Minneapolis, inner-city St. Paul, and inner-city South Minneapolis, respectively. Only 5% of the soil samples taken from the smaller urban areas of Rochester and St. Cloud, Minnesota, had Pb levels >150 μg/g. It has been suggested that the higher Pb levels associated with soils taken from around painted homes in the inner city are the result of greater atmospheric Pb content, resulting from the burning of leaded gasoline in cars and the washdown of building surfaces to which the small Pb particles adhere by rain (Mielke et al. 1989). A state-wide Minnesota study concluded that exterior Pb-based paint was the major source of contamination in severely contaminated soils located near the foundations of private residences and that aerosol Pb accounted for virtually all of the contamination found in soils removed from the influence of Pb-based paint. Contamination due to Pb-based paint was found to be “highly concentrated over a limited area, while contamination due to aerosol Pb was found to be less concentrated, but more widespread” (Schmitt et al. 1988).

Pb was analyzed in dust wipes and soil samples from 67 public housing projects containing 487 dwelling units across the United States (Succop et al. 2001). A total of 5,906 dust wipes and 1,222 soil samples were included in the data set. The median soil levels were 194 ppm near the foundation, 177 ppm near the walkways, and 145 ppm elsewhere in the yard. The maximum level, 3,900 ppm, was found in a foundation sample. Median dust Pb loading (μg m^{-2}) from kitchens, living rooms, and two children’s bedrooms were 151 (5th–95th percentile range: 22, 674), 936 (86, 10,190), and 8,560 (818, 313,000) for floor window sills and window troughs, respectively. Thirteen percent of the floor samples and 30% of the window sill samples from the rooms exceeded the HUD Interim Dust Lead Standards of 431 and 2,690 μg m^{-2} for floor and window sill samples, respectively.

5.5.4 Paint

Weathering and deterioration of Pb-based paint can contribute to the Pb content of dust and soil (Clark et al. 2004; Hunt et al. 1993; Jaeger et al. 1998; Lucas et al. 2014; Marcus and Elias 1995). A soil Pb study in Minneapolis, Minnesota, found that soil samples taken from around the foundations of homes with painted exteriors had a mean concentration of 522 μg/g, while soil samples taken from around the foundations of brick or stucco had a mean concentration of 158 μg/g (Schmitt et al. 1988). Pb-based paint, removed from surfaces by burning (gas torch or hot air gun), scraping, or sanding have been found to result, at least temporarily, in higher levels of exposure for families residing in these homes. A 2006 survey of U.S. housing stock estimated that 35% of 106 million housing units contained Pb-based paint and that approximately 20% of houses contained deteriorating Pb-based paint (HUD 2011).

5.5.5 Other Media

Pb has been detected in a variety of foods and spices (Lin et al. 2010). Pb may be introduced into food through uptake from soil into plants or atmospheric deposition onto plant surfaces, during transport to market, processing, and kitchen preparation (EPA 1986b). The ban on leaded gasoline as well as the use of welded (non-soldered) food cans during the 1980s are largely responsible for the decreases in levels of Pb in the U.S. diet beginning in the 1980s (FDA 2006). The FDA analyzed samples of foods commonly eaten by toddlers and infants for Pb and noted that levels of Pb in infant and toddler foods, on average, are relatively low (FDA 2016a). These results are summarized in Table 5-15. Selected data from the 2006–2011 FDA Total Diet Study Market Baskets are presented in Table 5-16 (FDA 2016b). Mean Pb levels in dairy products (e.g., milk, cheese, ice cream, cream, yogurt) were generally low or below the detection limit. The dairy product category with the highest Pb level was for low-fat fruit-flavored yogurt, with a mean concentration of 0.002 mg/kg for 24 analyses. Mean concentrations of Pb in fruits and vegetable were also generally low, with the highest concentrations in raisins (0.005 mg/kg), spinach (0.004 mg/kg), and lettuce (0.004 mg/kg). Mean concentration of Pb in baby foods ranged from not detected to 0.013 mg/kg. The highest levels reported were found in sweet potatoes (0.013 mg/kg), arrowroot cookies (0.012 mg/kg), grape juice (0.011 mg/kg), teething biscuits (0.008 mg/kg), and apple-cherry juice (0.008 mg/kg). Based on a multimedia Pb exposure modeling analysis for children 1–5 years old, below the 70th percentile of PbB in the general U.S. population, dietary intake was a major background exposure pathway (Zartarian et al. 2017)

Table 5-15. Lead Levels in Foods Commonly Eaten by Toddlers and Infants

Product category	Average[a] (range) (μg/kg)	Number of samples
Cereal, infant/toddler (rice)	15.6 (5.0–82.0)	76
Cereal, infant/toddler (multigrain)	7.2 (6.0–8.0)	6
Cereal, infant/toddler (non-rice)	4.8 (0.4–17.0)	30

Table 5-15. Lead Levels in Foods Commonly Eaten by Toddlers and Infants

Product category	Average[a] (range) (µg/kg)	Number of samples
Apples[b]	3.3	10
Cereal, oat ring	7.8 (3.3–16.4)	30
Grapes	3.7 (3.3–7.6)	10
Juice, grape	5.6 (0.3–41.3)	30
Juice boxes and pouches	3.3 (0.3–17.0)	40
Peanut butter	5.3 (3.3–45.2)	29
Quinoa	22.2 (0.4–98.0)	30
Raisins	18.1 (1.8–151)	23
Stage 2 toddler foods	5.2 (1.0–22.2)	35
Teething biscuits	12.0 (2.0–131)	27
Toddler puffs	19.1 (3.3-–91.0)	31

[a] The average concentration reported for each product category was calculated using all values. For those samples with results below the detection limit, half of the detection limit was used to calculate the average.
[b] All of the apple samples were below the limit of detection.

Source: FDA 2016a

Table 5-16. Selected Mean Lead Concentrations in Food from the FDA Total Diet Study

Food	Mean (range) (mg/kg)[a]	Number of analyses	Number <LOD	LOD (mg/kg)
Syrup, chocolate	0.016 (0–0.027)	24	1	0.007
Apricots, canned in heavy/light syrup	0.015 (0–0.036)	24	1	0.007
Baby food, sweet potatoes	0.013 (0–0.034)	24	5	0.007
Peach, canned in light/medium syrup	0.013 (0–0.038)	24	2	0.007
Candy bar, milk chocolate, plain	0.013 (0–0.027)	24	5	0.01
Baby food, arrowroot cookies	0.012 (0–0.031)	24	9	0.01
Sweet potatoes, canned	0.012 (0–0.018)	24	2	0.007
Shrimp, boiled	0.012 (0–0.18)	24	18	0.01
Baby food, juice, grape	0.011 (0–0.02)	24	1	0.004
Fruit cocktail, canned in light syrup	0.011 (0–0.025)	24	4	0.007
Brownie	0.01 (0–0.032)	24	5	0.007

[a] Note: 1 mg/kg = 1,000 µg/kg.

FDA = U.S. Food and Drug Administration; LOD = limit of detection

Source: FDA 2016b

The U.S. Fish and Wildlife Service reported the concentrations of metals in a total of 315 composite samples of whole fish sampled from 109 stations nationwide from late 1994 to early 1995. For Pb, the geometric mean, maximum, and 85th percentile concentrations (μg/g wet weight) were 0.11, 4.88, and 0.22, respectively. The mean concentration of Pb was significantly lower than in the 1980–1981 survey. Pb concentrations in fish have declined steadily from 1976 to 1984, suggesting that reductions of leaded gasoline and controls on mining and industrial discharges have reduced Pb in the aquatic environment (Schmitt and Brumbaugh 1990).

In order to reduce Pb exposure from consumption of Pb-contaminated fish and shellfish, consumption advisories are issued by states recommending that individuals restrict their consumption of specific fish and shellfish species from certain water bodies where Pb concentrations in fish and shellfish tissues exceed the human health level of concern. This level of concern is set by individual state agencies and used to issue advisories recommending no consumption, or restricted consumption, of contaminated fish and shellfish from certain waterbody types (e.g., lakes and/or rivers). In 1995, the EPA Office of Water issued guidance to states on sampling and analysis procedures to use in assessing the health risks from consuming locally caught fish and shellfish. The risk assessment method proposed by EPA was specifically designed to assist states in developing fish consumption advisories for recreational and subsistence fishers (EPA 1995a). These two groups within the general population consume larger quantities of fish and shellfish than the general population and frequently fish the same water bodies routinely. Because of this, these populations are at greater risk of exposure to Pb and other chemical contaminants if the waters they fish are contaminated. In 2007, eight advisories restricting the consumption of Pb-contaminated fish and shellfish were in effect in five states (Hawaii, Idaho, Washington, Kansas, and Missouri) and one territory (American Samoa) (EPA 2007b).

Elevated levels of Pb in the blood of cattle grazing near a Pb smelter have been reported, although no implications regarding Pb in beef were made. The mean Pb levels for the herd were highest near the smelter and decreased with distance. Ingestion of soil along with the forage was thought to be a large source of additional metal (Neuman and Dollhopf 1992). Evidence has also been shown for transfer of Pb to milk and edible tissue in cattle poisoned by licking the remains of storage batteries burned and left in a pasture (Oskarsson et al. 1992). Levels of Pb in muscle of acutely sick cows that were slaughtered ranged from 0.23 to 0.5 mg/kg (wet weight basis). Normal Pb levels in bovine meat from Swedish farms are <0.005 mg/kg. For eight cows that were less exposed, levels of Pb in milk taken 2 weeks after the exposure were 0.08±0.04 mg/kg. The highest Pb level found in the milk of eight cows studied for 18 weeks was 0.22 mg/kg. Pb in most milk samples decreased to values <0.03 mg/kg 6 weeks after exposure. Two affected cows delivered a calf at 35 and 38 weeks after the exposure. There was a high Pb level in the blood of the cows at the time of delivery, which suggests mobilization of Pb in connection with the latter stages of gestation and delivery. Pb levels in colostrum were increased as compared to mature milk samples taken 18 weeks after exposure. The concentration of Pb in milk produced after delivery decreased rapidly with time and was almost down to the limit of detection in mature milk.

In a survey, 324 multivitamin-mineral products were analyzed for Pb content (Mindak et al. 2008). Estimates of Pb exposure from these products were derived for four groups summarized in Table 5-17. The overall median value for Pb exposure was 0.576 µg/day. Five samples would have provided exposures that exceeded 4 µg/day. The authors reported that the estimates of Pb exposures were below the provisional total tolerable intake levels for the four population groups (Mindak et al. 2008). Twenty-one elements, including Pb, were analyzed in various botanical and dietary supplements; Pb concentrations ranged from not detected to 4.21 µg/g. None of the products analyzed would result in a maximum exposure that exceeds a tolerable level of exposure (Avula et al. 2010).

Table 5-17. Estimated Median and Maximum Lead Exposures

Population group	Median (µg/day)	Maximum (µg/day)
Young children (0–6 years)	0.123	2.88
Older children (7+ years)	0.356	1.78
Pregnant or lactating women	0.845	8.97
Adult women	0.842	4.92

Source: Adapted with permission from Mindak et al. (2008), American Chemical Society.

Many non-Western folk remedies used to treat diarrhea or other ailments may contain substantial amounts of Pb. Examples of these include: Alarcon, Ghasard, Alkohl, Greta, Azarcon, Liga, Bali Goli, Pay-loo-ah, Coral, and Rueda. In addition, an adult case of Pb poisoning was attributed to an Asian remedy for menstrual cramps known as Koo Sar. The pills contained Pb at levels as high as 12 ppm (CDC 1998). The source of the Pb was thought to be in the red dye used to color the pills. Pb was the most common heavy metal contaminant/adulterant found in samples (n=54) of Asian traditional remedies available at health food stores and Asian groceries in Florida, New York, and New Jersey (Garvey et al. 2001). Sixty percent of the remedies tested would give a daily dose of Pb in excess of 300 mg when taken according to labeling instructions. Pb poisoning has been caused by ingestion of a Chinese herbal medicine to which metallic Pb was added to increase its weight and sales price (Wu et al. 1996). Ayurveda is a traditional form of medicine practiced in India and other South Asian countries; the medications used often contain herbs, minerals, metals, or animal products and are made in standardized and nonstandardized formulations (CDC 2004b). CDC (1998, 2002b) reported cases of elevated PbBs in children after consuming candy from Mexico or using various folk remedies. Elevated PbBs were reported in two 7-year-old children in Rhode Island. A sample of litargirio, which was used as an antiperspirant/deodorant, found in the home contained 79% Pb (CDC 2005).

During 2011–2012, six cases of Pb poisoning were associated with the use of 10 oral Ayurvedic medications made in India. Pb concentrations in these medications were as high as 2.4%. Blood Pb levels of these women ranged from 16 to 64 μg/dL (CDC 2012c). In 2004–2012, the New York City Department of Health and Mental Hygiene identified 22 oral medications, supplements, or remedies containing high levels of heavy metals, including Pb (Table 5-18).

Table 5-18. Lead Content in Ayurvedic Medications and Other Health Remedies

Product	Country where manufactured	Country where purchased	Lead content (ppm)
Calabash Chalk (Nzu)	Unknown	United States	6.6
Emperor’s Tea Pill (concentrated)	China	United States	5,400
Garbha Chintamani Ras (Vrihat) (Swarna Yukt)	India	India	120
Garbha Dharak Yog	India	India	110
Garbhapal Ras	India	India	22,000
Garbhapal Ras	India	United States	15,000
Hepatico Extract (concentrated)	China	United States	5,900
Jambrulin	India	United States	243,000
Kankayan Bati (Gulma)	India	United States	12
Lakshmivilash Ras (Nardiya)	India	United States	260
Laxmana Louh	India	India	180
Maha Sudarshan	India	United States	41
Mahashakti Rasayan	India	India	9,400
Mahayogaraj Guggulu (enriched with silver)	India	United States	47,000
Ovarin	India	India	24,000
Pigmento	India	India	7.3
Pregnita	India	India	12,000
Sorin	India	India	46,707
Tierra Santa	Mexico	United States	13
Vasant Kusumakar Ras (with Gold and Pearl)	India	India	29
Vatvidhwansan Ras	India	United States	20,000
Vita Breath	United States	United States	1,100

Source: CDC 2012c

A study was conducted in an urban neighborhood in Chicago in order to gauge the levels of Pb in an array of fruits, vegetables, and herbs (Finster et al. 2004). The soil Pb concentrations where the plants were sampled varied from 27 to 4,580 ppm (median 800 ppm, geometric mean 639 ppm). Detectable Pb levels in the edible fruit, vegetables, and herbs sampled ranged from 11 to 81 ppm. Only one fruiting vegetable (cucumber 81 ppm) among the 52 sampled had detectable levels of Pb in the edible portion. However, 12 of the 31 leafy vegetables and herbs sampled contained Pb in the edible shoot part of the plant (range, 11–60 ppm). The Pb concentrations in the four samples of root vegetables ranged from 10 to 21 ppm. No significant correlation was found between the Pb concentrations in the edible portion of plant and the soil Pb level.

Pb may leach from Pb crystal decanters and glasses into the liquids they contain. Port wine that contained an initial concentration of 89 μg/L Pb was stored for 4 months in crystal decanters containing up to 32% Pb oxide. At the end of 4 months, Pb concentrations in the port were 5,331, 3,061, and 2,162 μg/L in decanters containing 32, 32, and 24% Pb oxide, respectively. Pb was also found to elute from Pb crystal wine glasses within minutes. Mean Pb concentrations in wine contained in 12 glasses rose from 33 μg/L initially to 68, 81, 92, and 99 μg/L after 1, 2, 3, and 4 hours, respectively (Graziano and Blum 1991).

Hair dyes and some cosmetics may contain Pb compounds (Cohen and Roe 1991). Hair dyes formulated with Pb acetate may have Pb concentrations 3–10 times the allowable concentration in paint. Measured Pb concentrations of 2,300–6,000 μg of Pb/gram of product have been reported (Mielke et al. 1997). Pb acetate is soluble in water and easily transferred to hands and other surfaces during and following application of a hair dye product. Measurements of 150–700 μg of Pb on each hand following application have been reported (Mielke et al. 1997). In addition to transfer of Pb to the hand-to-mouth pathway of the person applying the product, Pb can be transferred to any other surface (comb, hair dryer, outside of product container, counter top, etc.) that comes into contact with the product. It is also on the hair that it is applied to and the hands applying it. Objects coming into contact with hair dyed with a Pb-containing product also become contaminated. A dry hand passed through dry hair dyed with a Pb-containing product in cream form was been shown to pick up about 786 μg of Pb. A dry hand passed through dry hair dyed using foam or liquid Pb-containing hair dye products picked up less Pb: 69 μg/hand for foam products and 73 μg/hand for liquid products (Mielke et al. 1997). An elevated PbB (12 μg/dL) in an infant was observed after the use of tiro, a Nigerian eye cosmetic applied to the infant’s eyes (CDC 2012a). Elevated PbBs (27.0 and 33.5 μg/dL) were reported in two young children in New Mexico after the use of kajal, a cosmetic imported from Afghanistan, that was applied to the children’s eyelids. The kajal was reported to contain 54% Pb (CDC 2013). Sindoor, a cosmetic and cultural/religious powder used in Hindu cultures, has been found to contain very high amounts of Pb (Lin et al. 2010).

Cases of Pb poisoning have been related to less common sources of exposure. Illicit "moonshine" whiskey made in stills composed of Pb-soldered parts (e.g., truck radiators) may contain high levels of Pb. Detectable levels of Pb with a maximum concentration of 5.3 mg/L were found in 7 of 12 samples of Georgia moonshine whiskey (Gerhardt et al. 1980). Of the 115 suspected moonshine samples seized by local law enforcement between 1995 and 2001 and analyzed by the Bureau of Alcohol, Tobacco, and Firearms, 33 samples (28.7%) contained Pb levels >300 μg/dL. The median and maximum levels were 44.0 and 53,200 μg/dL, respectively (Parramore et al. 2001).

Firing of Pb ammunition may result in exposure to Pb aerosols and dusts generated during gun or rifle discharge at levels up to 1,000 $\mu g/m^3$ (EPA 1985c), from Pb pellets ingested by or imbedded in animals that are used as food sources, and from Pb pellets or fragments imbedded in humans from shooting incidents (see Appendix C, Ingestion of Lead Debris). Exposures to airborne Pb dust from firearm discharge in indoor shooting ranges has been shown to result in increases in PbBs that are 1.5–2 times higher than preexposure concentrations (Greenberg and Hamilton 1999; Gulson et al. 2002). However, the use of copper-jacketed bullets, nonlead primers, and well-ventilated indoor firing ranges lessen the impact of airborne Pb on blood Pb levels (Gulson et al. 2002).

A Pb poisoning hazard for young children exists in imported vinyl miniblinds that had Pb added to stabilize the plastic. Over time, the plastic deteriorates to produce Pb dust that can be ingested when the blinds are touched by children, who then put their hands in their mouths (CPSC 1996). The U.S. Consumer Product Safety Commission (CPSC) has requested that manufacturers change the manufacturing process to eliminate the Pb. As a consequence, vinyl miniblinds should now be Pb-free. The CPSC recommends that consumers with young children remove old vinyl miniblinds from their homes and replace them with new miniblinds made without added Pb or with alternative window coverings.

Inexpensive metallic jewelry items specifically intended for children and teenagers have been shown to contain varying levels of Pb (Maas et al. 2005). A total of 311 chemical assays conducted using 285 jewelry items purchased in 20 different stores in California revealed that a considerable amount of Pb was added to the items, presumably to increase their weight or to impart some type of metallic coating to the surface of the item. The mean weight percentage of Pb for all 311 assays was 30.6%. Of the

311 samples tested, 169 contained at least 3% Pb by weight in at least one portion of the jewelry piece and 123 of the samples were found to contain >50% Pb by weight (Maas et al. 2005). In addition, 62 pieces of the purchased jewelry were tested for surface levels of Pb that could potentially be transferred dermally through the routine handling of these pieces. Using standard laboratory wipes, the surface of the jewelry pieces were wiped for a total of 20 seconds and subsequently analyzed for Pb content. Mean Pb levels in the wipes ranged from 0.06 to 541.97 μg. The authors characterized the potential Pb exposure from these dermal transfer experiments as either low exposure (<1 μg of Pb transferred to the laboratory wipe), moderate exposure (1–10 μg of Pb transferred to the laboratory wipe), high exposure (10–50 μg of Pb transferred to the laboratory wipe), and very high exposure (>50 μg of Pb transferred to the laboratory wipe). Approximately 35% of the 62 pieces tested were characterized as having low exposure, 48% were characterized as moderate exposure, 11% were characterized as high exposure, and 5% were characterized as very high exposure (Maas et al. 2005).

5.6 GENERAL POPULATION EXPOSURE

Measurements of Pb in blood, urine, and tissues (postmortem) have been used to assess exposures of individuals to Pb. Table 5-19 shows the lowest limit of detections that are achieved by analytical analysis of blood, urine and tissues.

Table 5-19. Lowest Limit of Detection Based on Standards[a]

Media	Detection limit	Reference
Whole blood/urine/tissue	0.05 µg Pb/g blood or mL urine	NIOSH 1994b, Method 8003
	1 µg/100 g blood; 0.2 µg/g tissue	NIOSH 1994a, Method 8005
Animal tissue	0.1 µg/g (ICP-MS or GFAA)	NOAA 1998

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

GFAA = graphite furnace atomic absorption; ICP-MS = inductively coupled plasma-mass spectrometry

Prior to the 1980s, aerolized Pb emissions from the use of leaded gasoline was the main source of Pb exposure for the general U.S. population. Aerolized Pb can be either inhaled or ingested after deposition on surfaces and food crops. Adult Pb exposures tend to be limited to occupational or recreational sources. For children, the primary source of Pb exposure is from surface dusts (on the ground or entrained) that contain Pb from a variety of sources including deteriorated Pb-based paint (Bornschein et al. 1986; CDC 2009; Dixon et al. 2009; Egeghy et al. 2005; EPA 1996c; Garavan et al. 2008; Gulson et al. 2009;

Lanphear and Roghmann 1997; Lanphear et al. 1998a; Lewin et al. 1999; Malcoe et al. 2002; Mielke et al. 2007; Succop et al. 1998; Von Lindern et al. 2003, 2016; Zahran et al. 2013). Young children are particularly vulnerable to Pb exposure because of hand-to-mouth activity, which contributes to ingestion of Pb in surface dusts. Pb in the fine particle fraction of surface dusts (<150 μm) readily adheres to the skin surface, from which it can be inadvertently ingested from hand-to-mouth activity (Choate et al. 2006a, 2006b; Clausing et al. 1987; Davis and Mirick 2006; Davis et al. 1990; Siciliano et al. 2009; Yamamoto et al. 2006). Several studies have attempted to quantify soil and dust ingestion in children (Chien et al. 2017; Ozkaynak et al. 2011; Sedman et al. 1994; Stanek et al. 2012; Von Lindern et al. 2016; Wilson et al. 2013) and adults (Calabrese et al. 1990; Doyle et al. 2012; Irvine et al. 2014; Stanek et al. 1997).

Although air Pb can be a direct pathway of exposure in children, it can also be an indirect pathway from its effect on Pb concentration in surface dusts (Brunekreef 1984; Hayes et al. 1994; Hilts 2003; Rabinowitz et al. 1985; Schnaas et al. 2004; Schwartz and Pitcher 1989; Tripathi et al. 2001). Second-hand smoke may also contribute to increased Pb exposure (Apostolou et al. 2012; Mannino et al. 2003; Richter et al. 2013). Dietary sources of Pb can originate from direct or indirect transfer of atmospheric Pb emissions to secondary media such as water, food crops, game, and fish. Pb in the maternal system can also be transferred to the fetus during gestation and to the nursing infant (EPA 2014c).

Several studies provided data on Pb levels in food, with which dietary intakes of Pb for the general population in the United States have been estimated (FDA 2016a, 2016b). An analysis of individual food intakes and PbB from NHANES (2006–2008) estimated that diet explained approximately 2.9% of the variations of PbB in children and 1.6% in adults (Davis et al. 2014). A randomized survey of 250 individuals (adults and children) from the Midwest United States conducted over the period 1995–1997 estimated average dietary Pb intake to be approximately 10 μg/day (Clayton et al. 1999). The EPA has estimated mean dietary Pb intakes in children ages 6–84 months to be approximately 2 μg/day (EPA 2014c). The ban on the use of welded (non-soldered) food cans during the 1980s has resulted in a decrease in Pb exposure from foods (FDA 2006). In recent surveys, the mean Pb levels in dairy products (e.g., milk, cheese, ice cream, cream, yogurt) were generally low or below the detection limit. Mean concentrations of Pb in fruits and vegetables were also generally low. Mean concentration of Pb in baby foods ranged from not detected to 0.013 mg/kg. Possible sources of Pb in food samples include introduction during processing or preparation with drinking water contaminated with Pb, deposition of Pb onto raw materials for each food, and Pb exposure in livestock that produce dairy or meat ingredients (EPA 2014c). Pb has also been reported in home-prepared reconstituted infant formula. Although, at one time, use of Pb solder in formula containers contributed to PbB from formula consumption (Ryu et al. 1983), this practice was phased out after 1970 in the United States and subsequently banned (FDA 1995). However, tap water remains a potential source of Pb in home-prepared formula at locations where tap water Pb concentrations are elevated. In a study conducted in the Boston area in 1997, 2 of 40 samples of home-prepared formula had Pb concentrations >15 μg/L. In both cases, the reconstituted formula had been prepared using cold tap water run for 5–30 seconds, drawn from the plumbing of houses >20 years old. Pb-containing ceramic ware used in food preparation has also been associated with childhood Pb exposure in children of Hispanic ethnicity in San Diego County, California. One study (Gersberg et al. 1997) used the IEUBK Model to determine that dietary Pb exposure from beans prepared in Mexican ceramic bean pots may account for a major fraction of blood Pb burden in children whose families use such ceramic ware.

The main source of Pb in drinking water is from the corrosion of Pb service lines, which are pipes constructed of pure Pb that connect the water distribution main to a building's internal plumbing. Other common sources of Pb in drinking water are exposed leaded solder or corroded fixtures containing Pb (EPA 2016a). While Pb was restricted to no more than 8% in plumbing materials in 1986, older homes and neighborhoods may still contain Pb service lines, Pb connections, Pb solder, or other Pb-based plumbing materials that may contaminate drinking water during its delivery from its source to homes. Corrosion of these older plumbing materials can result in leaching of Pb into drinking water (CDC 2012b; Hanna-Attisha et al. 2016). Flint, Michigan is an example of how a water system with Pb sources in drinking water infrastructure resulted in elevated Pb levels in drinking water. For decades, the drinking water for the City of Flint was purchased from the Detroit Water and Sewer Department (DWSD). This water had optimized corrosion control and was treated with orthophosphate, a corrosion inhibitor that reduces Pb solubility and leaching from leaded plumbing materials by the formation of protective scales on the pipe's interior surface. When the water source was changed to the Flint River in 2014, corrosion control was not implemented, which allowed Pb to leach into the drinking water (EPA 2017e). Pb concentration in first-draw tap water tends to be higher than after the plumbing system has been flushed, although with Pb service lines, it is possible to see higher Pb concentrations in flushed water, if flushing is sufficient to draw stagnant water from the service line to the tap. Gulson et al. (1997a) measured Pb in household water throughout the day when the plumbing system of an unoccupied test house was not flushed. Water concentration data ranged from 119 μg/L for the initial (first-draw) sample to 35–52 μg/L for hourly samples to 1.7 μg/L for a fully flushed sample. The 1991 LCR was implemented to protect public health by minimizing Pb and copper levels in drinking water, by primarily reducing water corrosivity (EPA 2010). The rule set a Pb action level of 15 μg/L based on 90th percentile levels of tap water samples. The LCR established tap sampling monitoring requirements for public water systems. One-liter samples are taken at the tap where water has stood in the pipes for at least 6 hours (first-draw) in homes and buildings that are considered high-risk of Pb and copper contamination, and the number of samples are based on the system size. Pb action level exceedances can trigger a number of steps that a water system can take to reduce Pb exposure. These requirements include implementing a corrosion control treatment program, monitoring and/or treating source water, public education, and Pb service line replacement (EPA 2004). As discussed in Section 5.5.2, EPA has proposed major changes in the LCR as of October 2019.

Other less common sources of Pb exposure also exist. Exposure may also result from engaging in hobbies that use Pb (e.g., leaded solder is used in making stained glass, molten Pb used in casting, leaded glazes and frits are used in making pottery, and Pb compounds as coloring agents in glassblowing) (Grabo 1997). The use of inadequately glazed or heavily worn earthenware vessels for food storage and cooking may result in Pb exposure (CDC 1985; EPA 1986b). Various folk remedies and Ayurvedic medication (CDC 1998, 2004b, 2012c; Garvey et al. 2001; Wu et al. 1996) and some cosmetics (Mielke et al. 1997) may also be sources of Pb exposure. Moonshine consumption was strongly associated with elevated PbBs (Morgan and Parramore 2001). A 2000 study found a median PbB of 11 μg/dL among 35 moonshine consumers versus 2.5 μg/dL in 68 randomly-selected nonmoonshine consumers (Parramore et al. 2001). Exposure to infants and children can occur from mouthing of leaded jewelry and toys containing Pb or painted with leaded paint (CDC 2018c).

Plastic food wrappers may be printed with pigments that contain Pb chromates. Plastic wrappers used for 14 different national brands of bread collected in New Jersey contained a mean concentration of 26 mg of Pb for a bag size of 2,000 cm^2. A survey of 106 homemakers who buy such breads indicated that 39% of them reused the bags and 16% of the respondents turned the bags inside out to reuse them, suggesting that the potential exists for Pb leaching from the paint into the stored food (Weisel et al. 1991).

Blood Pb levels measured as a part of the NHANES revealed that between 1976 and 1991, the mean PbBs of the U.S. population aged 1–74 years old dropped 78%, from 12.8 to 2.8 μg/dL. The prevalence of PbBs ≥10 μg/dL also decreased sharply from 77.8 to 4.3%. The major cause of the observed decline in PbBs is most likely the removal of 99.8% of Pb from gasoline and the removal of Pb from soldered cans (Pirkle et al. 1994). Data from the Fourth National Report on Human Exposure to Environmental Chemical are summarized in Tables 5-20 and 5-21, which provide geometric means of Pb levels in the blood and urine in segments of the U.S. population.

Table 5-20. Geometric Mean Blood Lead Levels (µg/dL) and the 95th Percentile Confidence Interval, by Race/Ethnicity, Sex, and Age for the Years for 2011–2016

	Survey years	Geometric mean (95% confidence interval)	Sample size
Total	11–12	0.973 (0.916–1.04)	7,920
	13–14	0.858 (0.813–0.906)	5,215
	15–16	0.820 (0.772–0.872)	4,988
Age group			
1–5 years	11–12	0.970 (0.877–1.07)	713
	13–14	0.782 (0.705–0.869)	818
	15–16	0.758 (0.675–0.850	790
6–11 years	11–12	0.681 (0.623–0.744)	1,048
	13–14	0.567 (0.529–0.607)	1,075
	15–16	0.571 (0.523–0.623)	565
12–19 years	11–12	0.554 (0.511–0.601)	1,129
	13–14	0.506 (0.464–0.551)	627
	15–16	0.467 (0.433–0.504	1,023
20 years and older	11–12	1.09 (1.03–1.16)	5,030
	13–14	0.967 (0.921–1.02)	2,695
	15–16	0.920 (0.862–0.982)	2,610
Gender			
Males	11–12	1.13 (1.06–1.21)	3,968
	13–14	0.994 (0.919–1.08)	2,587
	15–16	1.13 (1.06–1.21)	3,968
Females	11–12	0.842 (0.796–0.890)	3,952
	13–14	0.746 (0.715–0.777)	2,628
	15–16	0.735 (0.679–0.795)	2,500
Race/ethnicity			
Mexican Americans	11–12	0.838 (0.767–0.916)	1,077
	13–14	0.746 (0.685–0.813)	969
	15–16	0.704 (0.659–0.759)	994
Non-Hispanic blacks	11–12	0.998 (0.947–1.05)	2,195
	13–14	0.871 (0.787–0.963)	1,119
	15–16	0.856 (0.763–0.962	1,070
Non-Hispanic whites	11–12	0.993 (0.914–1.08)	2,493
	13–14	0.882 (0.820–0.950)	1,848
	15–16	0.835 (0.774–0.900)	1,511
All Hispanics	11–12	0.855 (0.793–0.922)	1,931
	13–14	0.742 (0.695–0.793)	1,481
	15–16	0.703 (0.658–0.750)	1,664
Asians	11–12	1.15 (1.06–1.24)	1,005
	13–14	1.01 (0.923–1.11)	510

Table 5-20. Geometric Mean Blood Lead Levels (µg/dL) and the 95th Percentile Confidence Interval, by Race/Ethnicity, Sex, and Age for the Years for 2011–2016

	Survey years	Geometric mean (95% confidence interval)	Sample size
	15–16	1.07 (0.976–1.18)	479

Source: CDC 2018a

Table 5-21. Geometric Mean Urine Lead Levels (µg/dL) and the 95th Percentile Confidence Interval, by Race/Ethnicity, Sex, and Age

	Survey years	Geometric mean (95% confidence interval)	Sample size
Total	11–12	0.360 (0.328–0.396)	2,504
	13–14	0.277 (0.257–0.298)	2,664
	15–16		3,061
Age group			
3–5 years	15–16	0.257 (0.225–0.292)	486
6–11 years	11–12	0.346 (0.292–0.410)	399
	13–14	0.222 (0.192–0.258)	402
	15–16	0.346 (0.292–0.410)	399
12–19 years	11–12	0.259 (0.219–0.305)	390
	13–14	0.201 (0.166–0.245)	451
	15–16	0.196 (0.183–0.211)	402
20 years and older	11–12	0.381 (0.348–0.416)	1,715
	13–14	0.297 (0.280–0.315)	1,811
	15–16	0.304 (0.276–0.334)	1,794
Gender			
Males	11–12	0.414 (0.367–0.466)	1,262
	13–14	0.315 (0.295–0.337)	1,318
	15–16	0.313 (0.285–0.343)	1,524
Females	11–12	0.316 (0.282–0.355)	1,242
	13–14	0.245 (0.222–0.269)	1,346
	15–16	0.259 (0.233–0.288)	1,537
Race/ethnicity			
Mexican Americans	11–12	0.372 (0.320–0.431)	317
	13–14	0.277 (0.240–0.319)	453
	15–16	0.259 (0.233–0.288)	585
Non-Hispanic blacks	11–12	0.431 (0.385–0.483)	669
	13–14	0.371 (0.320–0.429)	581
		0.340 (0.298–0.388)	671
Non-Hispanic whites	11–12	0.346 (0.311–0.385)	820
	13–14	0.267 (0.245–0.290)	985
	15–16	0.275 (0.247–0.305)	924
All Hispanics	11–12	0.372 (0.327–0.423)	573

Table 5-21. Geometric Mean Urine Lead Levels (µg/dL) and the 95th Percentile Confidence Interval, by Race/Ethnicity, Sex, and Age

	Survey years	Geometric mean (95% confidence interval)	Sample size
	13–14	0.270 (0.239–0.305)	701
	15–16	0.284 (0.258–0.312)	982
Asians	11–12	0.383 (0.341–0.429)	353
	13–14	0.257 (0.230–0.287)	292
	15–16	0.292 (0.264–0.324)	332

Source: CDC 2019

The Adult Blood Lead Epidemiology and Surveillance (ABLES) program tracks adult (aged ≥16 years) cases with elevated PbBs from workplace exposure. In 2016, 26 states submitted PbB data on 18,093 adults with PbBs ≥10 µg/dL. PbBs ≥10 µg/dL declined from 26.6 adults per 100,000 employed in 2010 to 15.8 per 100,000 employed in 2016 (results for data submitted as of December 2018). In 2016, among adults with known exposures, 90.3% had occupational exposure. The majority of these adults were employed in manufacturing, construction, mining, and services. Table 5-22 presents industries within each sector with the most workers with occupational exposures resulting in PbB ≥25 µg/dL during 2010–2016 (NIOSH 2017a).

Table 5-22. Industries by Sector with Most Workers having Blood Lead Concentrations (PbBs) ≥25 µg/dL, 2010–2016

NORA Sector	Industry NAICS Code
Manufacturing	Storage battery manufacturing (33591)
	Nonferrous metal (except copper and aluminum) rolling, drawing, extruding, and alloying (33149)
	Alumina and aluminum production and processing (33131)
	Nonferrous metal foundries (33152)
	Nonferrous metal (except aluminum) smelting and refining (33141)
	Other basic inorganic chemical manufacturing (32518)
	Motor vehicle electrical and electronic equipment manufacturing (33632)
Construction	Painting and wall covering contractors (23832)
	Highway, street, and bridge construction (23731)
	Residential building construction (23611)
	Plumbing, heating, and air-conditioning contractors (23822)
	Site preparation contractors (23891)
	Commercial and institutional building construction (23622)
Services (except public safety)	All other amusement and recreation industries (71399)
	Remediation services (56291)

Table 5-22. Industries by Sector with Most Workers having Blood Lead Concentrations (PbBs) ≥25 µg/dL, 2010–2016

NORA Sector	Industry NAICS Code
	Automotive mechanical and electrical repair and maintenance (81111)
	Other services (except public safety industries) (71394)
Mining	Copper, nickel, lead, and zinc mining (21223)

NAICS = North American Industry Classification System; NORA = National Occupational Research Agenda

Source: NIOSH 2017a

Raymond and Brown (2015a, 2015b, 2017) and analyzed the 2007–2012 and 2009–2014 datasets from the Childhood Blood Lead Surveillance (CBLS) system. In 2007, a total of 38 states identified and reported 37,289 children (<6 years) with PbB ≥10 µg/dL. In 2012, a total of 30 jurisdictions identified and reported approximately 138,000 children (<6 years) with PbB ≥5 µg/dL. In 2012, federal funding ended and several states lost their state-wide Pb poisoning prevention programs and in 2013, the number of states reporting data declined, as did the number of children reported to the CDC with PbB ≥5 µg/dL. In October 2013, federal funding resumed and in 2013, 27 states, the District of Columbia, and New York City reported data. In 2014, 30 states, the District of Columbia, and New York City reported data. Table 5-23 summarizes the number and rate per 100,000 children aged <5 years with blood Pb levels 5–9 µg/dL reported in the 2010–2014 CBLS system. PbBs ≥10 µg/dL continue to be more prevalent among children with known risk factors, such as minority race or ethnicity, urban residence, residing in homes built prior to the 1950s, and low family income (CDC 2009).

Table 5-23. Number and Rate per 100,000 Children Aged <5 Years with Blood Lead Levels 5–9 µg/dL in the Childhood Blood Lead Surveillance System, United States, 2010–2014

	<1 Year		1–4 Years	
Year	Number	Rate	Number	Rate
2010[a]	18,598	448.48	137,887	805.62
2011[b]	13,981	352.69	130,838	810.56
2012[c]	7,876	199.74	95,854	596.58
2013[d]	5,494	138.26	57,293	360.46
2014[e]	5,904	148.51	70,680	444.49

[a] 37 jurisdictions reporting.
[b] 36 jurisdictions reporting.
[c] 30 jurisdictions reporting.
[d] 29 jurisdictions reporting.
[e] 32 jurisdictions reporting.

Table 5-23. Number and Rate per 100,000 Children Aged <5 Years with Blood Lead Levels 5–9 µg/dL in the Childhood Blood Lead Surveillance System, United States, 2010–2014

	<1 Year		1–4 Years	
Year	Number	Rate	Number	Rate

Source: Raymond and Brown 2017

Various studies suggest that ingestion of game hunted with Pb shot is associated with increased PbBs. Johansen et al. (2006) collected blood samples from 50 men in Nuuk, Greenland to study the relationship between the consumption of birds hunted with Pb shot and PbBs. Men who regularly ate hunted birds killed with Pb shot had mean PbB ranging from 6.2 µg/dL in the group eating 0.1–5 bird equivalents per month to 12.8 µg/dL in those eating >30 bird equivalents per month. In addition, levels were highest in mid-winter when consumption of hunted birds was highest. Those who did not consume hunted birds had a mean PbB of 1.5 µg/dL. These results are consistent with earlier surveys of Arctic hunting communities. A 1992 survey of 492 Inuit adults from the Arctic region of Quebec, Canada showed that consumption of waterfowl, along with age and smoking, were associated with elevated PbB (Dewailly et al. 2001). The geometric mean PbB was 0.42 µmol/L (8.7 µg/dL), with a range of 0.04–2.28 µmol/L (0.8–47 µg/dL). In a cohort of Inuit newborns from northern Quebec, where the population consumed game killed with Pb shot, the geometric umbilical cord PbB was 3.9 µg/dL (range 0.2–27 µg/dL); 7% of Inuit newborns had cord PbBs >10 µg/dL as compared to 0.16% of the non-Inuit population in southern Quebec (Lévesque et al. 2003).

Second-hand smoke may also contribute to increased Pb exposure (Apostolou et al. 2012; Mannino et al. 2003; Richter et al. 2013). Pb is a component of tobacco and tobacco smoke, and smokers often have higher Pb blood levels than nonsmokers (Bonanno et al. 2001; Mannino et al. 2003). Using data from the NHEXAS EPA Region V study, PbB levels in smokers and nonsmokers were analyzed and a correlation between tobacco smoke and exposure levels was observed (Bonanno et al. 2001). The mean PbBs in smokers, nonsmokers exposed to environmental tobacco smoke (ETS), and nonsmokers without ETS were 2.85, 2.06, and 1.81 µg/dL, respectively (Bonanno et al. 2001). Recent Pb urine concentrations for the U.S. adult population from the NHANES by smoking status are presented in Table 5-24.

Table 5-24. Geometric Mean Urine Lead Levels (μg/dL) and the 95th Percentile Confidence Interval by Smoking Status

	Survey years	Geometric mean (95% confidence interval)	Sample size
Cigarette smokers			
Total	11–12	2.36 (1.71–4.62)	876
	13–14	1.51 (1.30–1.91)	957
Age group			
20–49 years	11–12	1.78 (1.41–3.07)	522
18–49 years	13–14	1.34 (1.13–1.92)	583
50 years and older	11–12	3.35 (1.62–6.83)	354
	13–14	1.72 (1.40–2.03)	374
Gender			
Males	11–12	3.07 (1.73–5.03)	527
	13–14	1.91 (1.48–2.14)	512
Females	11–12	1.58 (1.14–3.45)	349
	13–14	1.30 (1.12–1.41)	445
Nonsmokers[a]			
Total	11–12	1.38 (1.25–1.58)	1,343
	13–14	1.16 (0.950–1.51)	1,487
Age group			
20–49 years	11–12	1.26 (1.02–1.38)	671
18–49 years	13–14	0.880 (0.720–1.04)	778
50 years and older	11–12	1.63 (1.29–2.16)	672
	13–14	1.48 (1.12–2.52)	709
Gender			
Males	11–12	1.61 (1.18–2.13)	635
	13–14	1.51 (1.04–2.68)	663
Females	11–12	1.32 (1.06–1.38)	708
	13–14	0.238 (0.219–0.258)	824

[a]Cigarette nonsmokers who used other tobacco products were excluded.

Source: CDC 2018a

Studies have been conducted to determine exposure of firearm instructors to Pb at outdoor firing ranges when either nonjacketed (pure Pb) or jacketed (copper-coated) bullets were used. Instructors are likely to have higher exposure than shooters because they spend more time at the range. In studies at an outdoor range in Virginia, the mean breathing zone Pb level when nonjacketed bullets were fired was 67.1 μg/m^3 for one instructor and 211.1 μg/m^3 for another (Tripathi and Llewellyn 1990). When jacketed bullets were used, breathing zone levels decreased to ≤8.7 μg/m^3. PbBs of the instructors did not exceed the OSHA Pb standard's medical removal level of 2.4 μmol/L (60 μg/dL) in either case (OSHA 2016a).

When shooters fired conventional Pb bullets, their mean exposures to airborne Pb were 128 μg/m^3 in the personal breathing zone and 68 μg/m^3 in the general area. When totally copper-jacketed Pb bullets were fired, the mean breathing zone and general area air sample concentrations were 9.53 and 5.80 μg/m^3, respectively (Tripathi and Llewellyn 1990). At an outdoor uncovered range in Los Angeles, instructors who spent an average of 15–20 hours/week behind the firing line were found to be exposed to breathing zone Pb concentrations of 460 and 510 μg/m^3 measured as 3-hour, time-weighted averages. The PbB of one instructor reached 3.38 μmol/L (70 μg/dL). After reassignment to other duties, repeat testing indicated his PbB had dropped to 1.35 μmol/L (28 μg/dL) (Goldberg et al. 1991).

In 1991, NIOSH conducted a survey of the Federal Bureau of Investigations (FBI) Firearms Training Unit firing ranges and related facilities to determine occupational Pb exposures among FBI and Drug Enforcement Agency (DEA) firing range personnel (NIOSH 1996b). Sixty-one personal breathing-zone and 30 area samples for airborne Pb were collected. Exposures ranged up to 51.7 μg/m^3 (mean, 12.4 μg/m^3), 2.7 μg/m^3 (mean, 0.6 μg/m^3), and 4.5 μg/m^3 (mean, 0.6 μg/m^3) for range instructors, technicians, and gunsmiths, respectively. Exposure of custodians ranged from nondetectable to 220 μg/m^3 during short-term cleaning of a large indoor range. Carpet dust sampling of dormitory rooms of students who practiced at the firing ranges revealed higher ($p<0.0005$) dust-Pb concentrations when compared to nonstudent dormitories (dust-Pb concentration range of 116–546 μg/g with a geometric mean of 214 μg/g in the student's rooms versus a dust-Pb concentration range of 50–188 μg/g with a geometric mean of 65 μg/g for the nonstudent rooms). This suggested that the students were contaminating their living quarters with Pb.

The American Academy of Pediatrics (AAP) (1998, 2005) concluded that although monitoring data demonstrate a decline in PbBs, Pb remains a common, preventable, environmental health threat. Most Pb poisoning in children is the result of dust and chips from deteriorating Pb paint on interior surfaces (AAP 2005, 2016; ATSDR 2017). The AAP supported the CDC guidelines endorsing universal screening in certain areas and targeted screening for children at high risk (CDC 1997b, 2005). Many children continue to be at risk for ingestion of Pb-based paint and of soil and dust contaminated through the deterioration of Pb-based paint and the residues from combustion of leaded gasoline. A 1974 study indicated that elevated PbBs in children were most likely a result of ingesting Pb-contaminated soil, and that the most likely source was Pb-based paint rather than Pb from automotive exhaust (Ter Haar and Aronow 1974). However, more recent studies have shown that children with the highest PbBs live in areas with high traffic flow where Pb particles in the air may fall directly to the soil or adhere to the outer surfaces of building and wash to the soil with rain (Mielke et al. 1989, 2008, 2010). The CDC concluded that a common source of Pb exposure for children who have elevated PbB is Pb-based paint that has deteriorated into paint chips and Pb dusts (CDC 1997b, 2012d).

Pb can readily cross the placenta; therefore, exposure of women to Pb during pregnancy results in uptake by the fetus. Furthermore, since the physiological stress of pregnancy may result in mobilization of Pb from maternal bone, fetal uptake of Pb can occur from a mother who was exposed to Pb before pregnancy, even if no Pb exposure occurs during pregnancy. Maternal Pb can also be transferred to breastfeeding infants.

Malcoe et al. (2002) assessed Pb sources and their effect on blood Pb in rural Native American and white children living in a former mining region. Blood samples, residential environmental samples (soil, dust, paint, water), and caregiver interviews (hand-mouth behaviors, socioeconomic conditions) were obtained from a representative sample of 245 children ages 1–6 years. There were no ethnic differences in the results. However, poor children were especially vulnerable. Regression analysis showed that mean floor dust Pb loading >10.1 μg/ft^2 and yard soil Pb >165.3 mg/kg were independently associated with blood Pb levels ≥10 μg/dL.

The Pb content of dusts can be a significant source of exposure, especially for young children. Baseline estimates of potential human exposure to dusts, including intake due to normal hand-to-mouth activity, are 0.2 g/day for children 1–6 years old versus 0.1 g/day for adults when both indoor and outdoor ingestion of soil including dust is considered (EPA 1989a). For children who engage in pica behavior (the compulsive, habitual consumption of nonfood items), the ingestion rate of soil can be as high as 5 g/day. Although ingestion of Pb-containing paint may lead to elevated PbBs in young children, a major source of elevated PbBs (>10 μg/dL) in children is often contaminated household dust and subsequent hand contamination and repetitive mouthing (Bornschein et al. 1986; Charney et al. 1980; Dixon et al. 2009; Lanphear and Roghmann 1997; Lanphear et al. 1998a; Succop et al. 1998). Weathering of Pb-based paint can contribute to the Pb content of dust and soil. Pb levels of indoor dust and outdoor soil were found to be strongly predictive of PbBs in over 200 urban and suburban infants followed from birth to 2 years of age; however, PbBs were not correlated with indoor air or tap water Pb levels, nor the size of nearby roadways. Indoor dust Pb levels and soil Pb levels in the homes of children with high PbBs (>8.8 μg/dL) were 72 μg/wipe (window sill dust) and 1,011 μg/g, respectively; children with low PbBs (<3.7 μg/dL) were exposed to 22 μg/wipe and 380 μg/g, respectively. In addition, 79% of the homes of children with high PbBs had been renovated, while only 56% of the homes of children with low PbBs had been renovated, suggesting that renovating the interior of homes previously painted with leaded paint may increase, at least temporarily, a child's exposure to Pb dust (Rabinowitz et al. 1985). Regular use of dust control methods (e.g., wet mopping of floors, damp-sponging of horizontal surfaces, high-efficiency vacuum cleaner) has been shown in some, although not all, cases to reduce indoor dust, Pb dust, and blood Pb levels in some, although not all, older homes containing leaded paints (Lanphear et al. 2000b; Rhoads et al. 1999). Decreases of between 17 and 43% in blood Pb concentrations were observed in children where regular dust control methods had been used to reduce indoor levels of Pb (Rhoads et al. 1999). EPA (2014c) summarized concentrations of Pb in house dust in the United Stated from 2006 to 2011; these data are presented in Table 5-25.

PbB samples from 1,473 children <5 years old were analyzed prior to and after the change in drinking water source in the city of Flint, Michigan (Hanna-Attisha et al. 2016). Prior to the change, 2.4% of the children had PbB levels exceeding 5 µg/dL (n=736). Following the change in water source, 4.9% of children’s PbB levels exceeded 5 µg/dL for samples obtained from January 1 to September 15, 2015 (n=737). The study also found that in areas where ≥25% of the drinking water samples exceeded 15 µg/L, the percentage of children with PbB levels >5 µg/dL increased from 4.0 to 10.6%. Gomez et al. (2018) analyzed PbB levels for children <5 years old in Flint, Michigan over an 11-year time span from 2006 to 2016. The percentage of children with PbB levels >5.0 µg/dL declined from 11.8% in 2006 to 3.2% by 2016. The study authors noted an uptick in the geometric mean PbB level during the height of the Flint water crisis from 1.19±0.02 to 1.30±0.02 µg/dL in 2014–2015, but it declined to 1.15±0.02 µg/dL in 2016 after the water source was switched back to the DWSD. The authors concluded that while there was a slight increase in PbB levels during the time at which the source of drinking water was changed for residents of Flint, the overall trend for the 11-year time span was decreasing PbB levels with a nearly 73% reduction in the percentage of children having levels >5 µg/dL. A second study analyzed PbB levels for females aged 12–50 years prior to (April 25, 2012–October 15, 2013), during (April 25, 2014–October 15, 2015), and immediately after (April 25, 2016–October 15, 2017) the Flint water crisis (Gomez et al. 2019). The authors found that blood levels did not increase for females of child-bearing age residing in Flint during the period when the water supply was changed from the DWSD to the Flint River. The geometric means reported were 0.69 µg/dL (April 25, 2012–October 15, 2013), 0.65 µg/dL (April 25, 2014–October 15, 2015), and 0.55 µg/dL, (April 25, 2016–October 15, 2017).

Table 5-25. Measurements of Lead in Indoor Dust in the United States from 2006 to 2011

Location	Sample site	Value reported
New York City, New York	Glass plate next to open window of academic building	Median weekly dust loading: 52 μg/m^2
Eureka, Utah near Eureka Mills Superfund Site	Indoor home site (not specified)	Dust concentrations, range: 160–2,000 mg/kg
Denver, Colorado, near Vasquez Blvd and I-70 Superfund Site	Indoor home site (not specified)	Dust concentrations, range: 11–660 mg/kg
East Helena, Montana, near East Helena Superfund Site	Indoor home site (not specified)	Dust concentrations, range: 68–1,000 mg/kg
Syracuse, New York	Floor	Dust concentrations, range: 209–1,770 mg/kg
United States (nationwide)	Smooth floor	Median dust loading: 1.7 μg/m^2 Average dust loading: 4.4 μg/m^2
	Rough floor	Median dust loading: 5.6 μg/m^2 Average dust loading: 16 μg/m^2
	Smooth windowsill	Median dust loading: 2.5 μg/m^2 Average dust loading: 190 μg/m^2
	Rough windowsill	Median dust loading: 55 μg/m^2 Average dust loading: 480 μg/m^2
Milwaukee, Wisconsin	Central perimeter	Average dust concentration: 107 μg/m^2
	Entry	Average dust concentration: 140 μg/m^2
	Window	Average dust concentration: 151 μg/m^2
Rural towns, Idaho	Vacuum	Dust concentration Median: 120 mg/kg Maximum: 830 mg/kg
	Floor	Median dust concentration: 95 mg/kg Maximum dust concentration: 1,300 mg/kg
Bunker Hill, Idaho Superfund Site	Vacuum	Median dust concentration: 470 mg/kg Maximum dust concentration: 2,000 mg/kg
	Floor	Median dust concentration: 290 mg/kg Maximum dust concentration: 4,600 mg/kg

Source: EPA 2014c

Lanphear and Roghmann (1997) and Lanphear et al. (1996a, 1996b, 1998b) studied factors affecting PbBs in urban children and found the following independent predictors of children's PbBs: dust Pb loading in homes (carpets, uncarpeted floors, window sills, and troughs), African-American race/ethnicity, foundation perimeter soil Pb levels, ingestion of soil or dirt, Pb content and condition of interior painted surfaces, and first-flush kitchen drinking water Pb levels (Lanphear et al. 1996a, 1996b). Differences in housing conditions and exposures to Pb-containing house dust appear to contribute to the racial differences in urban children's PbBs. In addition, white children were more likely to put soil in their mouths (outdoor exposure) and suck their fingers, and African-American children were more likely to put their mouths on window sills (indoor exposure) and to use a bottle. Interior Pb exposures were more significant for African American children and exterior Pb exposures were more significant for white children (Lanphear et al. 1996a, 1996b). Mouthing behaviors are an important mechanism of Pb exposure among urban children (Lanphear and Roghmann 1997). Community characteristics such as residence within a city, proportion of African Americans, lower housing value, housing built before 1950, higher population density, higher rates of poverty, lower percent of high school graduates, and lower rates of owner-occupied housing have been used to identify children with elevated blood levels (Lanphear et al. 1998b). An analysis of children's PbBs and multiple measures of Pb concentrations in household dust, tap water, foundation perimeter soil, and interior house paint has been used to predict the effect of changing concentrations of Pb in environmental media on children's PbBs. An increase in dust Pb loading from background to 200 μg/ft^2 was estimated to produce an increase of 23.3% in the percentage of children estimated to have a PbB >10 μg/dL; an increase in tap water Pb concentration from background to 15 μg/L was estimated to produce an increase of 13.7% in the percentage of children estimated to have a PbB level >10 μg/dL; and an increase in soil Pb concentration from background to 400 μg/g was estimated to produce an increase of 11.6% in the percentage of children estimated to have a PbB level >10 μg/dL (Lanphear et al. 1998a).

Outdoor Pb dust was found to be a more potent contaminant of children's hands than indoor dust at daycare centers in New Orleans; boys, in general, had higher hand Pb levels than girls. The conclusions were based on Pb analysis of hand wipe samples taken before and after children played outdoors at four different daycare centers (a private inner-city site, a private outer-city site, a public inner-city site, and a public outer-city site). The private inner-city site had a severely contaminated outdoor play area with measured soil Pb concentrations ranging from 287 to 1,878 mg/kg. The outdoor play area at the public inner-city site, where children exhibited the lowest hand Pb measurements of any site in the study, had been completely paved over with concrete or rubberized asphalt and had well-maintained equipment (Viverette et al. 1996).

EPA conducted the Urban Soil Lead Abatement Demonstration Project (USLADP), also known as the “Three City Lead Study,” in Boston, Baltimore, and Cincinnati (EPA 1996c). The purpose was to determine whether abatement of Pb in soil could reduce PbBs of inner-city children. No significant evidence was found that soil abatement had any direct impact on children’s PbBs in either the Baltimore or Cincinnati studies. In the Boston study, however, a mean soil Pb reduction of 1,856 ppm resulted in a mean decline of 1.28 μg/dL PbB at 11 months postabatement (Weitzman et al. 1993). Phase II extended the study to 2 years and included soil abatement of the two comparison areas from Phase I (Aschengrau et al. 1994). Combined results from Phase I and II suggested a higher impact of soil remediation on PbBs (2.2–2.7 μg/dL). EPA reanalyzed the data from the USLADP in an integrated report (EPA 1996c). They concluded that when soil is a significant source of Pb in the child's environment, under certain conditions, the abatement of that soil will result in a reduction in exposure and consequently, PbB level. The results of the USLADP suggest that a number of factors are important in determining the influence of soil remediation on PbBs in children. These include the site-specific exposure scenario, the magnitude of the remediation, and the magnitude of additional sources of Pb exposure.

Authors of a study of PbBs in children in Toronto, Canada, before and after abatement of Pb-contaminated soil and house dust found that they could neither strongly support nor refute beneficial effects of abatement. The failure to reach a definite conclusion from the results of the study, which included data from 12 cross-sectional blood-screening surveys that were conducted over an 8-year period, was due, in part, to a low response rate (32–75%) to questionnaires used to determine behavioral, household, lifestyle, neighborhood, and environmental factors relating to study participants (Langlois et al. 1996).

Seasonal variations in PbBs in children have been observed in a number of studies (Gulson et al. 2008; Haley and Talbot 2004; Havlena et al. 2009; Kemp et al. 2007; Johnson and Bretsch 2002; Johnson et al. 1996; Laidlaw et al. 2005; Yiin et al. 2000). These studies suggest a general trend of increasing PbB during late summer and early fall. In addition to seasonal patterns in behavior (e.g., outdoor activities), seasonal patterns in weather (humidity and wind velocity) that promote re-entrainment and transport of dust Pb may contribute to the observed seasonal patterns in PbB (Laidlaw et al. 2005, 2012).

In addition to the ingestion of hand soil/dust through normal hand-to-mouth activity, some children engage in pica behavior (consumption of nonfood items), which can put them at increased risk through ingestion of large amounts of soil contaminated with Pb. It has been estimated that an average child may ingest between 20 and 50 mg of soil/day and that a pica child may ingest ≥5,000 mg of soil/day (LaGoy 1987; Mielke et al. 1989). If the soil contains 100 μg/g of Pb, an average child may be exposed to 5 μg Pb/day from this source alone (Mielke et al. 1989), and a pica child may be exposed to >100 times that amount.

Improper removal of Pb from housing known to contain Pb-based paint can significantly increase Pb levels in dust, thus causing Pb toxicity in children living in the home during the Pb-removal process. Four such cases have been documented (Amitai et al. 1987). In January 1995, the New York State Department of Health identified 320 children in 258 households in New York State (excluding New York City) with PbBs ≥20 μg/dL that were considered to be attributable to residential renovation and remodeling (CDC 1997a).

Workers occupationally exposed to Pb can carry Pb home on clothing, bodies, or tools (take home exposure). PbBs of children in households of occupationally exposed workers were almost twice those of children in neighboring homes whose parents were not occupationally exposed to Pb (median ranges were 10–14 and 5–8 μg/dL, respectively) (Grandjean and Bach 1986). Young children (<6 years old) of workers exposed to high levels of Pb in workplace air at an electronic components plant (61–1,700 μg Pb/m^3 ambient concentrations) had significantly elevated PbBs (13.4 μg/dL) compared with children from the same locale whose parents did not work in the electronics plant (7.1 μg/dL) (Kaye et al. 1987). Based upon data collected from 1987 to 1994, children aged 1–5 years (n=139) of workers whose occupation resulted in Pb exposure had a geometric mean PbB of 9.3 μg/dL as compared to a U.S. population geometric mean of 3.6 μg/dL (Roscoe et al. 1999). Of this group, 52% of the children had PbBs ≥10 μg/dL compared to 8.9% of the U.S. population and 21% had PbBs ≥20 μg/dL compared to 1.1% of the U.S. population (Roscoe et al. 1999). However, improved industrial hygiene procedures are likely to have decreased worker take-home exposures. Exposures of Pb workers' families have been identified in nearly 30 different industries and occupations. Industries in which exposure of family members has been reported most often include Pb smelting, battery manufacturing and recycling, radiator repair, electrical components manufacturing, pottery and ceramics, and stained glass making (NIOSH 1995). Children of Pb-exposed construction workers may also be at increased risk (Whelan et al. 1997).

Children may be exposed to Pb because of activities associated with certain hobbies and artistic activities practiced by adults in the home. Some of the more obvious hobbies and activities involving use of Pb-containing materials include casting, stained glass, pottery, painting, glassblowing, and screenprinting. Activities involving use of Pb-containing materials should always be done in an area well-ventilated with outdoor air and should never be done with children in the same room or in close proximity. Maas et al. (2005) indicated that high levels of Pb are prevalent in inexpensive cosmetic jewelry that is sold to the general public at retail stores.

Accidental or intentional ingestion of folk remedies (e.g., Chinese herbal medicines and Ayurvedic medicines containing Pb) or use of the Pb containing eye cosmetic tiro in children (discussed in Section 5.5.5) represents another source for potential Pb-poisoning in children. Sindoor, a cosmetic and cultural/religious powder used in Hindu cultures, has been found to contain very high amounts of Pb (Lin et al. 2010). Hair dyes formulated with Pb acetate represent a potential source for Pb-poisoning both by accidental ingestion and by hand-to-mouth activity following contact with Pb-contaminated surfaces, including dyed hair of adults (Mielke et al. 1997).

Children may be exposed to Pb through the inhalation of second-hand smoke. Mannino et al. (2003) employed data from the NHANES III and analyzed PbBs of children aged 4–16 years who were exposed to high, low, and intermediate levels of second-hand smoke. Serum levels of the nicotine biomarker cotinine were used to classify the children into one of the three second-hand smoke exposure categories. The geometric mean PbBs were 1.5, 1.9, and 2.6 μg/dL for children with low (≤0.050–0.104 ng/mL), intermediate (0.105–0.562 ng/mL), and high (0.563–14.9 ng/mL) serum cotinine levels, respectively (Mannino et al. 2003).

5.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

In addition to workers exposed to Pb in the workplace and family members of workers exposed via take home exposure, other population groups are at risk for potential exposure to high levels of Pb. These include populations residing in older housing or buildings that contain deteriorating leaded paint or that have galvanized pipes, Pb service lines, or scales that contain Pb within a distribution public water system; in high-traffic areas with legacies from leaded gasoline; near sites where Pb was produced or disposed; or near one of the NPL hazardous waste sites where Pb has been detected in some environmental media (ATSDR 2017b; EPA 2014c, 2016a). Since Pb is often detected in tobacco and tobacco smoke, persons who use chewing tobacco or smoke or are exposed to second-hand smoke, may have higher PbB levels than persons that do not use these products (Apostolou et al. 2012; Bonanno et al. 2001; Richter et al. 2013). Recent studies have also found e-cigarettes to be a potential source of Pb exposure (Olmedo et al. 2018). Other Pb sources that can contribute to elevated exposures to individual children or adults include mouthing or ingestion of toys containing Pb and consumption of candy and folk remedies and illicitly manufactured drugs that contain Pb (CDC 2002b, 2018c).

General population exposure is most likely to occur through the ingestion of food and water contaminated with Pb. Based on a multimedia Pb exposure modeling analysis for children 1–5 years old at upper percentiles of PbB in the U.S. population, soil and dust ingestion are dominant exposure pathways, but for lower percentiles, other age groups (e.g., younger children), or specific local U.S. locations, the main exposure source/pathway could be different (Zartarian et al. 2017). However, some individuals and families may be exposed to additional sources of Pb in their homes. This is particularly true of older homes that may contain Pb-based paint. In an attempt to reduce the amount of exposure due to deteriorating leaded paint, the paint is commonly removed from homes by burning (gas torch or hot air gun), scraping, or sanding. These activities have been found to result, at least temporarily, in higher levels of exposure for families residing in these homes. In addition, those individuals involved in the paint removal process (i.e., do-it-yourself renovators and professionals who remove Pb) can be exposed to such excessive levels that Pb poisoning may occur (Chisolm 1986; Fischbein et al. 1981; Rabinowitz et al. 1985). Special populations at risk of high exposure to tetraethyl Pb include workers at hazardous waste sites and those involved in the manufacture and dispensing of tetraethyl Pb (Bress and Bidanset 1991).