

6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

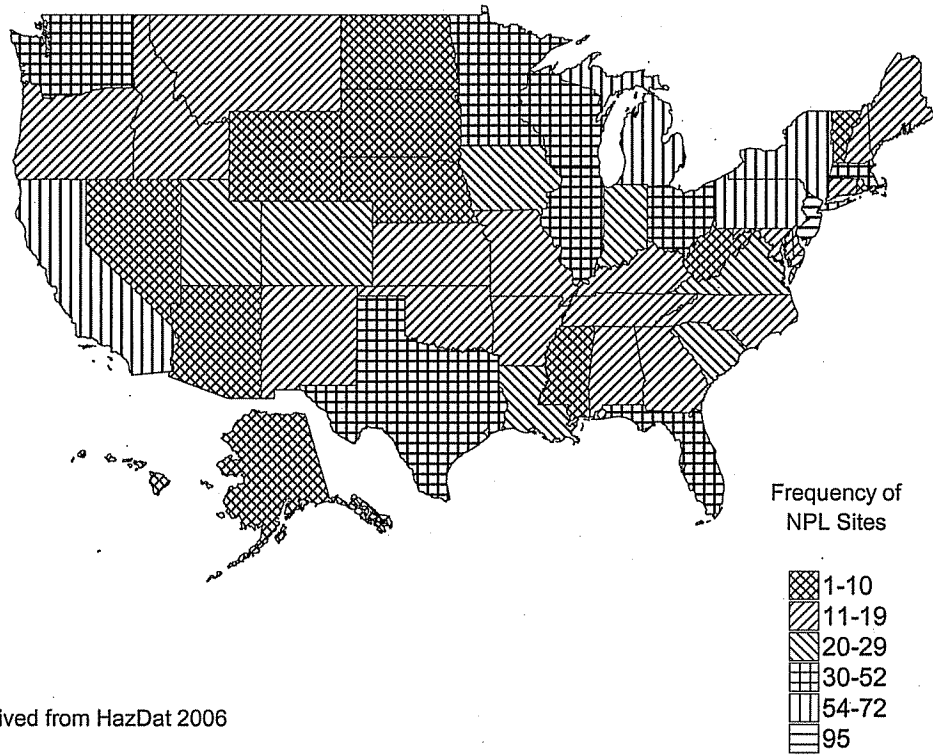
Arsenic has been identified in at least 1,149 of the 1,684 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2006). However, the number of sites evaluated for arsenic is not known. The frequency of these sites can be seen in Figure 6-1. Of these sites, 1,134 are located within the United States and 11, 2, and 2 are located in the Commonwealth of Puerto Rico, the Virgin Islands, and Guam (not shown).

Arsenic is widely distributed in the Earth's crust, which contains about 3.4 ppm arsenic (Wedepohl 1991). It is mostly found in nature in minerals, such as realgar (As_4S_4), orpiment (As_2S_3), and arsenolite (As_2O_3), and only found in its elemental form to a small extent. There are over 150 arsenic-bearing minerals (Budavari et al. 2001; Carapella 1992). While arsenic is released to the environment from natural sources such as wind-blown soil and volcanoes, releases from anthropogenic sources far exceed those from natural sources. Anthropogenic sources of arsenic include nonferrous metal mining and smelting, pesticide application, coal combustion, wood combustion, and waste incineration. Most anthropogenic releases of arsenic are to land or soil, primarily in the form of pesticides or solid wastes. However, substantial amounts are also released to air and water.

Arsenic found in soil either naturally occurring or from anthropogenic releases forms insoluble complexes with iron, aluminum, and magnesium oxides found in soil surfaces, and in this form, arsenic is relatively immobile. However, under reducing conditions, arsenic can be released from the solid phase, resulting in soluble mobile forms of arsenic, which may potentially leach into groundwater or result in runoff of arsenic into surface waters. In aquatic systems, inorganic arsenic occurs primarily in two oxidation states, As(V) and As(III). Both forms generally co-exist, although As(V) predominates under oxidizing conditions and As(III) predominates under reducing conditions. Arsenic may undergo a variety of reactions in the environment, including oxidation-reduction reactions, ligand exchange, precipitation, and biotransformation (EPA 1979, 1984a; Pongratz 1998; Welch et al. 1988). These reactions are influenced by Eh (the oxidation-reduction potential), pH, metal sulfide and sulfide ion concentrations, iron concentration, temperature, salinity, and distribution and composition of the biota (EPA 1979; Wakao et al. 1988). Much of the arsenic will adsorb to particulate matter and sediment. Arsenic released to air exists mainly in the form of particulate matter. Arsenic released from combustion processes will

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Figure 6-1. Frequency of NPL Sites with Arsenic Contamination



Derived from HazDat 2006

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generally occur as highly soluble oxides. These particles are dispersed by the wind and returned to the earth in wet or dry deposition. Arsines that are released to the atmosphere as a result of microbial action are oxidized to nonvolatile species that settle back to the ground.

Because arsenic is a natural component of the Earth's crust, low levels of the element are found in all environmental media. Atmospheric levels of arsenic in remote locations (away from human releases) range from 1 to 3 ng/m³, while concentrations in urban areas may range from 20 to 100 ng/m³. Concentrations in water are usually <10 µg/L, although higher levels may occur near natural mineral deposits or anthropogenic sources. Natural levels of arsenic in soil usually range from 1 to 40 mg/kg, with a mean of 5 mg/kg, although much higher levels may occur in mining areas, at waste sites, near high geological deposits of arsenic-rich minerals, or from pesticide application. Arsenic is also found in many foods, at concentrations that usually range from 20 to 140 µg/kg. Total arsenic concentrations may be substantially higher in certain seafoods. However, the general consensus in the literature is that about 85–>90% of the arsenic in the edible parts of marine fish and shellfish is organic arsenic (e.g., arsenobetaine, arsenocholine, dimethylarsinic acid) and that approximately 10% is inorganic arsenic (EPA 2003b). Drinking water in the United States generally contains an average of 2 µg/L of arsenic (EPA 1982c), although 12% of water supplies from surface water sources in the north Central region of the United States and 12% of supplies from groundwater sources in the western region have levels exceeding 20 µg/L (Karagas et al. 1998). In January 2001, EPA adopted a new standard that arsenic levels in drinking water were not to exceed 10 µg/L, replacing the previous standard of 50 µg/L. The date for compliance with the new MCL was January 23, 2006 (EPA 2001).

For most people, diet is the largest source of exposure to arsenic. Mean dietary intakes of total arsenic of 50.6 µg/day (range of 1.01–1,081 µg/day) and 58.5 µg/day (range of 0.21–1,276 µg/day) has been reported for females and males (MacIntosh et al. 1997). U.S. dietary intake of inorganic arsenic has been estimated to range from 1 to 20 µg/day, with grains and produce expected to be significant contributors to dietary inorganic arsenic intake (Schoof et al. 1999a, 1999b). The predominant dietary source of arsenic is generally seafood. Inorganic arsenic in seafood sampled in a market basket survey of inorganic arsenic in food ranged from <0.001 to 0.002 µg/g (Schoof et al. 1999a, 1999b). Intake of arsenic from air and soil are usually much smaller than that from food and water (Meacher et al. 2002).

People who produce or use arsenic compounds in occupations such as nonferrous metal smelting, pesticide manufacturing or application, wood preservation, semiconductor manufacturing, or glass production may be exposed to substantially higher levels of arsenic, mainly from dusts or aerosols in air.

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Exposure at waste sites may occur by a variety of pathways, including inhalation of dusts in air, ingestion of contaminated soil or water, or through the food chain. The magnitude of the exposures can only be evaluated on a site-by-site basis; however, exposures generally do not exceed background intakes from food and drinking water.

Tables 4-1, 4-2, 4-3, and 4-4 summarize all of the names, abbreviations, and structures of the various arsenic compounds that are discussed in Chapter 6.

6.2 RELEASES TO THE ENVIRONMENT

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

6.2.1 Air

Estimated releases of 4,800 pounds (~2.2 metric tons) of arsenic to the atmosphere from 58 domestic manufacturing and processing facilities in 2004, accounted for about 0.52% of the estimated total environmental releases from facilities required to report to the TRI (TRI04 2006). Estimated releases of 0.13 million pounds (~59 metric tons) of arsenic compounds to the atmosphere from 361 domestic manufacturing and processing facilities in 2004, accounted for about 0.11% of the estimated total environmental releases from facilities required to report to the TRI (TRI04 2006). These releases for arsenic and arsenic compounds are summarized in Table 6-1 and 6-2, respectively.

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Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Arsenic^a

State ^c	RF ^d	Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	Total release		
							On-site ^j	Off-site ^k	On- and off-site
AL	1	51	162	0	110,264	0	110,425	52	110,477
AR	2	0	0	No data	0	0	No data	0	0
AZ	2	10	0	0	20,717	0	20,727	0	20,727
CA	3	13	14	0	5,482	0	13	5,497	5,510
FL	2	4	0	0	0	4,950	4	4,950	4,954
GA	4	8	10	0	1,603	5	13	1,613	1,626
IA	1	0	1	0	0	0	0	1	1
ID	1	39	0	0	361,252	0	361,291	0	361,291
IL	2	250	129	0	14,087	0	379	14,087	14,466
IN	1	5	5	0	13,250	250	5	13,505	13,510
KS	1	0	0	No data	0	0	No data	0	0
KY	1	0	1	0	0	6	1	6	7
MI	2	0	5	0	0	750	5	750	755
MN	1	15	47	0	14,504	0	15	14,551	14,566
MO	1	5	0	0	0	4,040	5	4,040	4,045
MS	2	0	0	0	0	0	0	0	0
NC	4	35	8	0	1	1	43	2	45
NV	1	0	0	0	0	0	0	0	0
NY	4	0	1	0	26,525	1	26,401	126	26,527
OH	2	13	0	0	0	0	13	0	13
OR	1	0	0	0	92,606	0	92,606	0	92,606
PA	5	166	8	0	14,362	26,140	199	40,477	40,676
SC	3	10	10	0	0	1,002	15	1,007	1,022
TN	3	3,988	0	0	0	0	3,988	0	3,988
TX	5	139	376	168,563	12,600	0	181,636	42	181,678

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Table 6-1. Releases to the Environment from Facilities that Produce, Process, or Use Arsenic^a

State ^c	RF ^d	Reported amounts released in pounds per year ^b							Total release	
		Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	On-site ^j	Off-site ^k	On- and off-site	
WI	2	15	0	0	760	0	15	760	776	
WV	1	0	0	0	10,135	0	10,135	0	10,135	
Total	58	4,766	778	168,563	698,149	37,145	807,935	101,466	909,401	

^aThe 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.

^bData in TRI are maximum amounts released by each facility.

^cPost office state abbreviations are used.

^dNumber of reporting facilities.

^eThe sum of fugitive and point source releases are included in releases to air by a given facility.

^fSurface water discharges, waste water treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).

^gClass I wells, Class II-V wells, and underground injection.

^hResource Conservation and Recovery Act (RCRA) subtitle C landfills; other on-site landfills, land treatment, surface impoundments, other land disposal, other landfills.

ⁱStorage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown

^jThe sum of all releases of the chemical to air, land, water, and underground injection wells.

^kTotal amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI04 2006 (Data are from 2004)

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Table 6-2. Releases to the Environment from Facilities that Produce, Process, or Use Arsenic Compounds^a

State ^c	RF ^d	Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	Reported amounts released in pounds per year ^b		
							Total release		
							On-site ^j	Off-site ^k	On- and off-site
AK	1	511	0	1,400,000	1,200,000	0	2,600,511	0	2,600,511
AL	19	4,299	18,127	0	853,469	7,555	875,876	7,574	883,450
AR	12	0	0	0	133	26,435	0	26,568	26,568
AZ	5	5,421	0	0	402,335	422	394,749	13,429	408,178
CA	5	65	14	0	355,660	86,396	160,673	281,461	442,134
CO	1	11	0	0	4,094	0	4,105	0	4,105
CT	1	0	0	0	0	0	No data	0	0
FL	15	3,208	503	0	343,508	4,057	346,310	4,966	351,276
GA	23	8,643	7,823	0	422,124	5,127	437,496	6,221	443,717
HI	1	0	0	0	0	0	No data	0	0
IA	4	1,291	482	0	0	35,324	1,773	35,324	37,097
ID	3	332	20	0	1,056,904	0	1,057,256	0	1,057,256
IL	11	3,960	3,110	0	96,093	21,038	71,819	52,382	124,202
IN	21	13,786	8,282	0	768,297	42,808	632,704	200,470	833,174
KS	4	924	0	0	12,082	1	13,006	1	13,007
KY	18	14,406	8,427	0	616,074	95,285	578,080	156,112	734,192
LA	7	265	23	0	25,426	0	25,563	151	25,714
MA	1	0	0	0	0	500	0	500	500
MD	8	1,870	291	0	34,130	114,115	2,661	147,745	150,406
MI	10	1,123	2,310	68,924	101,857	1,059	77,505	97,769	175,274
MN	2	10	130	0	19,270	0	19,410	0	19,410
MO	6	462	116	0	27,855	936	10,026	19,343	29,369
MS	6	61	121	0	11,676	46	11,228	676	11,904
MT	3	630	0	0	2,138,190	37	2,138,820	37	2,138,857
NC	15	5,626	4,732	0	168,030	2,429	178,388	2,429	180,818
ND	6	6,326	5	0	318,175	0	137,961	186,545	324,506
NE	2	180	0	0	11,000	0	11,180	0	11,180
NJ	2	0	1	0	0	8	0	9	9
NM	2	130	0	0	18,326	0	18,456	0	18,456
NV	10	3,041	30,017	0	98,894,564	0	98,927,328	294	98,927,622
NY	3	67	36	0	27,059	802	27,141	823	27,964
OH	17	8,595	8,352	81,024	741,730	274	668,157	171,818	839,975
OK	4	115	13	0	25,000	4,202	115	29,215	29,330
OR	4	0	5	0	0	4,012	5	4,012	4,017
PA	23	18,963	2,166	0	666,753	69,053	403,582	353,353	756,935
PR	3	0	0	0	0	0	No data	0	0
RI	1	0	8	0	0	1,006	8	1,006	1,014

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Table 6-2. Releases to the Environment from Facilities that Produce, Process, or Use Arsenic Compounds^a

State ^c	RF ^d	Air ^e	Reported amounts released in pounds per year ^b							Total release	
			Water ^f	UI ^g	Land ^h	Other ⁱ	On-site ^j	Off-site ^k	On- and off-site		
SC	13	2,178	1,443	0	25,817	22,705	29,438	22,705	52,143		
SD	1	0	0	0	0	0	No data	0	0		
TN	13	3,379	25,878	0	292,914	17,219	258,643	80,746	339,389		
TX	17	4,616	199	33,148	196,385	31,557	226,751	39,155	265,906		
UT	5	6,715	4,500	0	6,368,500	3,500	6,379,715	3,500	6,383,215		
VA	11	1,911	2,773	0	160,154	8,463	164,789	8,512	173,301		
WA	4	0	0	0	0	0	No data	0	0		
WI	4	94	21	0	1,313	9,216	223	10,421	10,644		
WV	12	2,693	2,417	0	536,628	10,000	441,237	110,501	551,738		
WY	2	3,300	0	0	10,800	0	14,100	0	14,100		
Total	361	129,205	132,347	1,583,096	116,952,326	625,588	117,346,787	2,075,775	119,422,562		

^aThe 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.

^bData in TRI are maximum amounts released by each facility.

^cPost office state abbreviations are used.

^dNumber of reporting facilities.

^eThe sum of fugitive and point source releases are included in releases to air by a given facility.

^fSurface water discharges, waste water treatment-(metals only), and publicly owned treatment works (POTWs) (metal and metal compounds).

^gClass I wells, Class II-V wells, and underground injection.

^hResource Conservation and Recovery Act (RCRA) subtitle C landfills; other on-site landfills, land treatment, surface impoundments, other land disposal, other landfills.

ⁱStorage only, solidification/stabilization (metals only), other off-site management, transfers to waste broker for disposal, unknown

^jThe sum of all releases of the chemical to air, land, water, and underground injection wells.

^kTotal amount of chemical transferred off-site, including to POTWs.

RF = reporting facilities; UI = underground injection

Source: TRI04 2006 (Data are from 2004)

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Arsenic naturally occurs in soil and will be present in the atmosphere as airborne dust. It is also emitted from volcanoes and in areas of dormant volcanism (e.g., fumaroles). Gaseous alkyl arsenic compounds may be released from soil that has been treated with inorganic arsenic compounds as a result of biogenic processes (Schroeder et al. 1987; Tamaki and Frankenberger 1992). Arsenic naturally occurs in sea water and vegetation and is released into the atmosphere in sea salt spray and forest fires. Anthropogenic sources of arsenic include nonferrous metal smelting, coal, oil and wood combustion, and municipal waste incineration. Arsenic naturally occurs in coal and oil and therefore, coal- and oil-fired power plants release arsenic to the atmosphere in their emissions (Pacyna 1987). Arsenic's use in agriculture and industrial processes also contributes to its emissions. One important source of arsenic emissions is cotton ginning in which the cotton seeds are removed from the raw cotton.

The National Air Toxics Assessment reported that total anthropogenic emissions for arsenic compounds in the United States in 1996 were 355 tons/year (EPA 2005b). EPA conducted a modeling study with the Assessment System for Population Exposure Nationwide (ASPEN) in which estimates of emissions of hazardous air pollutants were used to estimate air quality (Rosenbaum et al. 1999). Using 1990 data, the total emissions of arsenic in the conterminous 48 states, excluding road dust or windblown dust from construction or agricultural tilling was estimated to be 3.0 tons/day with 90% of emissions coming from point sources and 5% each from area and mobile sources. U.S. emissions of arsenic to the atmosphere were estimated as 3,300 metric tons per year between 1979 and 1986 (Pacyna et al. 1995). There is evidence that anthropogenic emissions, at least from smelters, are lower than they had been in the early 1980s. It is likely that air releases of arsenic decreased during the 1980s due to regulations on industrial emissions (EPA 1986f), improved control technology for coal-burning facilities, and decreased use of arsenical pesticides.

Nriagu and Pacyna (1988) and Pacyna et al. (1995) estimated worldwide emissions of arsenic to the atmosphere for 1983. Estimates of yearly emissions from anthropogenic sources ranged from 12,000 to 25,600 metric tons with a median value of 18,800 metric tons. Natural sources contributed 1,100–23,500 metric tons annually. Chilvers and Peterson (1987) estimated global natural and anthropogenic arsenic emissions to the atmosphere as 73,500 and 28,100 metric tons per year, respectively. Copper smelting and coal combustion accounted for 65% of anthropogenic emissions. A U.S. Bureau of Mines study on the flow of mineral commodities estimated that global emissions of arsenic from metal smelting, coal burning, and other industrial uses ranged from 24,000 to 124,000 metric tons per year compared to natural releases, mostly from volcanoes, ranging from 2,800 to 8,000 metric tons per year (Loebenstein 1994).

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Pirrone and Keeler (1996) compared trends of trace element emissions from major anthropogenic sources in the Great Lakes region with ambient concentrations observed in urban areas of the region. They found that arsenic emissions increased about 2.8% per year from 1982 to 1988 and then decreased steadily by about 1.4% per year to 1993. Coal combustion in electric utilities and in residential, commercial, and industrial facilities was an important source of arsenic in the region, accounting for about 69% of the total emissions. Iron-steel manufacturing accounted for about 13% of the region wide arsenic emissions and nonferrous metals production for 17%.

Arsenic in the particulate phase is the predominant (89–98.6%) form of arsenic in the troposphere (Matschullat 2000). Inorganic species, most commonly trivalent arsenic, is the dominant form of arsenic in the air over emission areas; methylated forms of arsenic are probably of minor significance. Arsenic-containing air samples of smelter or coal-fired power plant origin consist largely of trivalent arsenic in both vapor and particulate form (Pacyna 1987). Oxides are the primary species evolved from fossil fuel and industrial processes. Additionally, arsenic trisulfide has also been reported from coal combustion, organic arsines from oil combustion, and arsenic trichloride from refuse incineration.

Arsenic has been identified in 59 air samples collected from 1,684 current or former NPL hazardous waste sites where it was detected in some environmental media (HazDat 2006).

6.2.2 Water

Estimated releases of 780 pounds (~0.35 metric tons) of arsenic to surface water from 58 domestic manufacturing and processing facilities in 2004, accounted for about 0.09% of the estimated total environmental releases from facilities required to report to the TRI (TRI04 2006). Estimated releases of 1.3×10^5 pounds (~59 metric tons) of arsenic compounds to surface water from 361 domestic manufacturing and processing facilities in 2004, accounted for about 0.11% of the estimated total environmental releases from facilities required to report to the TRI (TRI04 2006). These releases for arsenic and arsenic compounds are summarized in Tables 6-1 and 6-2, respectively.

Arsenic may be released to water from the natural weathering of soil and rocks, and in areas of vulcanism. Arsenic may also leach from soil and minerals into groundwater. Anthropogenic sources of arsenic releases to water include mining, nonferrous metals, especially copper, smelting, waste water, dumping of sewage sludge, coal burning power plants, manufacturing processes, urban runoff, atmospheric deposition and poultry farms (Garbarino et al. 2003; Nriagu and Pacyna 1988; Pacyna et al. 1995). A contributory

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part of mining and coal burning power plants is leaching from abandoned mine tailing and fly ash waste piles. Significant amounts of arsenic are released in liquid effluents from gold-milling operations using cyanide (Environment Canada 1993). Nriagu and Pacyna (1988) and Pacyna et al. (1995) estimated global anthropogenic inputs of arsenic into rivers, lakes, and oceans for 1983; annual estimated inputs ranged from 11,600 to 70,300 metric tons with a median value of 41,800 metric tons. Arsenic was detected in 58% of samples of urban storm water runoff from 8 of 15 cities surveyed in the National Urban Runoff Program at concentrations ranging from 1 to 50.5 µg/L (Cole et al. 1984).

Leaching of arsenic from soil, landfills, or slag deposits is a source of arsenic in groundwater (Francis and White 1987; Wadge and Hutton 1987). The arsenic in soil may be naturally-occurring or a result of the application of arsenic-containing pesticides or sludge. Wood treated with CCA is used in piers, piling and bulkheads and arsenic can leach from the treated wood (Breslin and Adler-Ivanbrook 1998; Brooks 1996; Cooper 1991; Sanders et al. 1994; Weis et al. 1998). Ammoniacal copper zinc arsenate (ACZA) is another arsenic-containing waterborne preservative; however, it is not as widely used as CCA (Lebow et al. 2000).

Arsenic has been identified in 846 groundwater and 414 surface water samples collected from 1,684 NPL hazardous waste sites, where it was detected in some environmental media (HazDat 2006).

6.2.3 Soil

Estimated releases of 0.70 million pounds (~320 metric tons) of arsenic to soils from 58 domestic manufacturing and processing facilities in 2004, accounted for about 77% of the estimated total environmental releases from facilities required to report to the TRI (TRI04 2006). An additional 0.17 million pounds (~77 metric tons), constituting about 19% of the total environmental emissions, were released via underground injection (TRI04 2006). Estimated releases of 117 million pounds (~5.3x10⁴ metric tons) of arsenic compounds to soils from 361 domestic manufacturing and processing facilities in 2004, accounted for about 98% of the estimated total environmental releases from facilities required to report to the TRI (TRI04 2006). An additional 1.6 million pounds (~720 metric tons), constituting about 1.3% of the total environmental emissions, were released via underground injection (TRI04 2006). These releases for arsenic and arsenic compounds are summarized in Tables 6-1 and 6-2, respectively.

The soil receives arsenic from a variety of anthropogenic sources, including ash residue from power plants, smelting operations, mining wastes, and municipal, commercial, and industrial waste. Ash from

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power plants is often incorporated into cement and other materials that are used for roads and construction. Arsenic may be released from such material into soil. Nriagu and Pacyna (1988) and Pacyna et al. (1995) estimated global anthropogenic inputs of arsenic into soil for 1983. Excluding mine tailings and smelter slag, annual estimated inputs ranged from 52,000 to 112,000 metric tons with a median value of 82,000 metric tons. Mine tailings and smelter slag were estimated to add an additional 7,200–11,000 and 4,500–9,000 metric tons, respectively. Old abandoned mine tailings undoubtedly contribute still more. Wood treated with CCA used in foundations or posts could potentially release arsenic into the surrounding soil. CCA preservatives have been shown to leach to varying degrees from wood, as well as through soils in both field and laboratory studies (Chirenje et al. 2003a; Hingston et al. 2001; Lebow et al. 2000; Rahman et al. 2004; Stilwell and Graetz 2001; USDA/USDT 2000). Arsenic may also be released on land through the application of pesticides and fertilizer. Senesi et al. (1999) reported the range of arsenic in 32 fertilizers as 2.2–322 ng/g. Roxarsone (3-nitro-4-hydroxyphenyl-arsonic acid), which was used to treat poultry feed in approximately 70% of the broiler poultry operations in 1999–2000, is excreted unchanged in the manure. Poultry litter (manure and bedding) is routinely used as fertilizer to cropland and pasture. In 2000, assuming 70% of the 8.3 billion broiler poultry produced in the United States were fed roxarsone-treated feed, the resulting manure would contain approximately 2.5×10^5 kg of arsenic (Garbarino et al. 2003). Land application of sewage sludge is another source of arsenic in soil. Arsenic was detected in sewage sludge samples from 23 cities at concentrations of 0.3–53 $\mu\text{g/g}$ (Mumma et al. 1984).

Arsenic has been identified in 758 soil and 515 sediment samples collected from 1,684 NPL hazardous waste sites, where it was detected in some environmental media (HazDat 2006).

6.3 ENVIRONMENTAL FATE

6.3.1 Transport and Partitioning

Arsenic in soil may be transported by wind or in runoff or may leach into the subsurface soil. However, because many arsenic compounds tend to partition to soil or sediment under oxidizing conditions, leaching usually does not transport arsenic to any great depth (EPA 1982c; Moore et al. 1988; Pantsar-Kallio and Manninen 1997; Welch et al. 1988). Arsenic is largely immobile in agricultural soils; therefore, it tends to concentrate and remain in upper soil layers indefinitely. Downward migration has been shown to be greater in a sandy soil than in a clay loam (Sanok et al. 1995). Arsenic from lead arsenate that was used for pest control did not migrate downward below 20 cm in one fruit orchard; in another orchard, 15 years after sludge amendments and deep plowing, essentially all arsenic residues

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remained in the upper 40 cm of soil (Merwin et al. 1994). Leaching of arsenic in polluted wetland soil was low; leaching was correlated with the amount of dissolved organic matter in the soil (Kalbitz and Wennrich 1998). The effect of soil characteristics, namely pH, organic matter content, clay content, iron oxide content, aluminum oxide content, and cation exchange capacity (CEC), on the adsorption of various metals, including the metalloid arsenic, to 20 Dutch surface soils was assessed by regression analysis (Janssen et al. 1997). The most influential parameter affecting arsenic adsorption was the iron content of the soil.

Arsenic that is adsorbed to iron and manganese oxides may be released under reducing conditions, which may occur in sediment or flooding conditions (LaForce et al. 1998; McGeehan 1996; Mok and Wai 1994). In addition to reductive dissolution, when nutrient levels are adequate, microbial action can also result in dissolution (LaForce et al. 1998). Interestingly, drying of the previously flooded soil increases arsenic adsorption, possibly due to alterations in iron mineralogy (McGeehan et al. 1998).

Darland and Inskeep (1997) conducted a study to determine the effects of pH and phosphate competition on the transport of arsenate ($H_xAsO_4^{x-3}$) through saturated columns filled with sand containing free iron oxides. At pH 4.5 and 6.5, arsenate transport was strongly retarded, while at pH 8.5, it was rapid. The enhanced transport of arsenate at pH 8 is consistent with the pH dependence of surface complexation reactions describing arsenate sorption by metal oxide minerals that can be categorized as a ligand exchange mechanism. Phosphate was shown to compete effectively with arsenate for adsorption sites on the sand, but the competition was not sufficient to desorb all of the arsenate in batch column experiments, even when the applied phosphate exceeded the column adsorption capacity by a factor of two. The researchers concluded that arsenate desorption kinetics may play an important role in the transport of arsenate through porous media. In a study looking at the effect of competing anions on the adsorption of arsenite and arsenate on ferrihydrite, the effect of phosphate on arsenate adsorption was greater at higher pH than at low pH and the opposite trend was observed for arsenite. While sulfate did not change the affinity of arsenate for ferrihydrite, sulfate reduced the adsorption of arsenite at pHs below 7.0 (Jain and Loeppert 2000).

Smith et al. (1999) investigated the sorption properties of both As(V) and As(III) in 10 Australian soils of widely different chemistry and mineralogy at commonly found arsenic levels. Adsorption of both arsenate and arsenite was rapid (1 hour). The amount of As(V) sorbed varied widely (1.7–62.0 L/kg); soils with lower amounts of oxidic material adsorbed much less arsenic than those with higher amounts of these minerals. Arsenate sorption was highly correlated with the iron oxide content of the soil and this

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factor probably accounts for much of the variation in soil adsorptivity. Considerable leaching of arsenic occurred at a separate site where cattle were treated with a dip containing arsenic (cattle dip site) and that contained similar soil properties to that studied by Smith et al. (1999). Arsenite adsorption, which was investigated in four of the Australian soils, was sorbed to a lesser extent than was arsenate. This was attributed to soil mineralogy and the species of As(V) (arsenate) and As(III) (arsenite) present in solution; at pH 5–7, the dominant As(V) species are H_2AsO_4^- and HAsO_4^{2-} and neutral H_3AsO_3 is the dominant As(III) species. For soils containing low amounts of oxidic minerals, pH had little effect on As(V) sorption, while for oxidic soils, a decrease in sorption was evident as the pH increased. In contrast, As(III) sorption increased with increasing pH (Smith et al. 1999). Jain et al. (1999) reported similar results where arsenite were both found to bind strongly to iron oxides; however, the adsorption of arsenate decreases with increasing pH, while the adsorption of arsenite increases with increasing pH (Jain et al. 1999). As(III), which exists in a neutral form as arsenous acid, H_3AsO_3 ($\text{pK}_a=9.23, 12.13, 13.4$), is less strongly adsorbed on mineral surfaces than the oxyanions of arsenic acid, H_3AsO_4 , ($\text{pK}_a=2.22, 6.98, 11.53$) (NRC 1999). Based on its pK_a values, arsenic acid would exist as a mixture of arsenate anions, H_2AsO_4^- and HAsO_4^{2-} , under most environmental conditions (pH 5–9).

The practice of liming to remediate contaminated soils and mine tailings has the potential to mobilize arsenic. Experiments performed by Jones et al. (1997) indicate that the increased mobility appears to be consistent with the pH dependence of sorption reactions of arsenic on iron oxide minerals rather than dissolution-precipitation reactions involving arsenic. They recommend that remediation of acidic mine tailings or other arsenic-contaminated soils be carefully evaluated with respect to potential arsenic mobilization, especially at contaminated sites hydraulically connected to surface or groundwaters.

Transport and partitioning of arsenic in water depends upon the chemical form (oxidation state and counter ion) of the arsenic and on interactions with other materials present. Soluble forms move with the water, and may be carried long distances through rivers (EPA 1979). However, arsenic may be adsorbed from water onto sediments or soils, especially clays, iron oxides, aluminum hydroxides, manganese compounds, and organic material (EPA 1979, 1982c; Welch et al. 1988). Under oxidizing and mildly reducing conditions, groundwater arsenic concentrations are usually controlled by adsorption rather than by mineral precipitation. The extent of arsenic adsorption under equilibrium conditions is characterized by the distribution coefficient, K_d , which measures the equilibrium partitioning ratio of adsorbed to dissolved contaminant. The value of K_d depends strongly upon the pH of the water, the arsenic oxidation state, and the temperature. In acidic and neutral waters, As(V) is extensively adsorbed, while As(III) is relatively weakly adsorbed. Trivalent inorganic arsenic exists predominantly as arsenous acid (H_3AsO_3)

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at environmental pH and is not strongly adsorbed to suspended solids and sediments in the water column. Pentavalent inorganic arsenic exists predominantly as H_2AsO_4^- and HAsO_4^{2-} in most environmental waters, which has considerably greater adsorption characteristics than arsenous acid. While in acidic and neutral waters, As(V) is more strongly adsorbed relative to As(III), in high-pH waters (pH >9) aquifer K_d values are considerably lower for both oxidation states (Mariner et al. 1996). Sediment-bound arsenic may be released back into the water by chemical or biological interconversions of arsenic species (see Section 6.3.2).

Arsenic enters rivers from where mining operations occurred and is transported downstream, moving from water and sediment into biofilm (attached algae, bacterial, and associated fine detrital material), and then into invertebrates and fish. The source of arsenic in the water column may be resuspended sediment. While arsenic bioaccumulates in animals, it does not appear to biomagnify between trophic levels (Eisler 1994; Farag et al. 1998; Williams et al. 2006).

Most anthropogenic arsenic emitted to the atmosphere arises from high temperature processes (e.g., coal and oil combustion, smelting operations, and refuse incineration) and occurs as fine particles with a mass median diameter of about 1 μm (Coles et al. 1979; Pacyna 1987). These particles are transported by wind and air currents until they are returned to earth by wet or dry deposition. Their residence time in the atmosphere is about 7–9 days, in which time the particles may be transported thousands of kilometers (EPA 1982b; Pacyna 1987). Long-range transport was evident in analyzing deposition of arsenic in countries like Norway; there was no indication that the marine environment contributed significantly to the deposition (Steinnes et al. 1992). Atmospheric fallout can be a significant source of arsenic in coastal and inland waters near industrial areas. Scudlark et al. (1994) determined the average wet depositional flux of arsenic as 49 $\mu\text{g As/m}^2/\text{year}$ for two sites in Chesapeake Bay, Maryland from June 1990 to July 1991. They found a high degree of spatial and temporal variability. The elemental fluxes derived predominantly from anthropogenic sources. Golomb et al. (1997) report average total (wet + dry) deposition rates to Massachusetts Bay of 132 $\mu\text{g/m}^2/\text{year}$, of which 21 $\mu\text{g/m}^2/\text{year}$ was wet deposition during the period September 15, 1992–September 16, 1993. Hoff et al. (1996) estimated the following arsenic loadings into the Great Lakes for 1994 (lake, wet deposition, dry deposition): Superior, 11,000 kg/year, 3,600 kg/year; Michigan, 5,000 kg/year, 1,800 kg/year; Erie, 5,500 kg/year, 1,800 kg/year; and Ontario, 3,000 kg/year, 580 kg/year. The measured dry deposition fluxes of arsenic at four sampling sites around Lake Michigan ranged approximately from 0.01 to 1.5 $\mu\text{g As/m}^2/\text{day}$; estimated inputs of arsenic into Lake Michigan were reported to be 1.4×10^3 kg/year (Shahin et al. 2000).

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Terrestrial plants may accumulate arsenic by root uptake from the soil or by absorption of airborne arsenic deposited on the leaves, and certain species may accumulate substantial levels (EPA 1982b). Yet, even when grown on highly polluted soil or soil naturally high in arsenic, the arsenic level taken up by the plants is comparatively low (Gebel et al. 1998b; Pitten et al. 1999). Kale, lettuce, carrots, and potatoes were grown in experimental plots surrounding a wood preservation factory in Denmark where waste wood was incinerated to investigate the amount and pathways for arsenic uptake by plants (Larsen et al. 1992). On incineration, the arsenate in the wood preservative was partially converted to arsenite; the arsenic emitted from the stack was primarily particle bound. Elevated levels of inorganic arsenic were found in the test plants and in the soil around the factory. Statistical analyses revealed that the dominating pathway for transport of arsenic from the factory to the leafy vegetables (kale) was by direct atmospheric deposition, while arsenic in the root crops (potatoes and carrots) was a result of both soil uptake and atmospheric deposition. Arsenic accumulation by plants is affected by arsenic speciation. Uptake of four arsenic species (arsenite, arsenate, methylarsonic acid, and dimethylarsinic acid) by turnips grown under soilless culture conditions showed that while uptake increased with increasing arsenic concentration in the nutrient, the organic arsenicals showed higher upward translocation than the inorganic arsenical (Carbonell-Barrachina et al. 1999). The total amount of arsenic taken up by the turnip plants (roots and shoots) followed the trend methylarsenate (MMA) < dimethylarsinic acid (DMA) < arsenite < arsenate. In a similar experiment, conducted with tomato plants, the total amount of arsenic taken up by the tomato plants followed the trend DMA < MMA < arsenate ≈ arsenite, with arsenic concentrations in the plants increasing with increasing arsenic concentration in the nutrient solution. Arsenic was mainly accumulated in the root system (85%) with smaller amounts translocating to the fruit (1%). However, plants treated with MMA and DMA had higher arsenic concentrations in the shoots and fruit than those treated with arsenite or arsenate (Burlo et al. 1999). Terrestrial plants growing on land bordering arsenic-contaminated waters show relatively little arsenic content, even though the sediments have arsenic concentrations as high as 200 µg/g (Tamaki and Frankenberger 1992). Arsenic concentrations in vegetables grown in uncontaminated soils and contaminated soils containing arsenic, as well as other metals and organic contaminants, were generally < 12 µg/kg wet weight. A maximum arsenic concentration of 18 µg/kg wet weight was found in unpeeled carrots grown in soil, which contained a mean arsenic concentration of 27 mg/kg dry weight (Samsøe-Petersen et al. 2002).

In a study by Rahman et al. (2004), CCA-treated lumber was used to construct raised garden beds to determine how far the components of CCA migrated in the soil and the uptake of these components by crops grown in the soil. Arsenic was found to diffuse laterally into the soil from the CCA-treated wood, with the highest concentrations found at 0–2 cm from the treated wood and a steady decline in

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concentration with increased distance. The highest average arsenic concentrations found in soil closest (0–2 cm) to the CCA-treated wood were 56 and 46 $\mu\text{g/g}$ in loamy sand and sandy loam soils, respectively. At a distance of 30–35 cm from the CCA-treated wood, arsenic concentrations were approximately 7 $\mu\text{g/g}$ in both soils. All samples were of the top 0–15 cm of soil. Crops grown in both soil types within 0–2 cm of the CCA-treated wood contained higher concentrations of arsenic, 0.186 and 10.894 $\mu\text{g/g}$ for carrots without peel and bean leaves and stems, respectively, than those grown at 1.5 m from the CCA-treated wood, 0.006 and 0.682 $\mu\text{g/g}$ for bean pods and bean leaves and stems, respectively. However, based on FDA guidelines on tolerance limits, these crops would be considered approved for human consumption. Studies by Chirenje et al. (2003a) also showed that elevated arsenic concentrations were found in surface (0–5 cm) soils immediately surrounding, within the first 0.3 m, of utility poles, fences, and decks made with CCA-treated wood. Factors such as the preservative formula, fixation temperature, post treatment handling, and timber dimensions of CCA-treated wood, as well as the pH, salinity, and temperature of the leaching media can affect the leach rates from CCA-treated wood (Hingston et al. 2001). Studies of leaching of the components of CCA- and ACZA-treated wood used to construct a boardwalk in wetland environments reported elevated arsenic levels in soil and sediment below and adjacent to these structures. Generally, these levels decreased with increasing distance from the structure (Lebow et al. 2000). Increased concentrations of arsenic were also observed under CCA-treated bridges. Arsenic levels declined with distance from the bridge and were near background levels at 1.8–3 m from the bridge's perimeter (USDA/USDT 2000).

In a study by Lebow et al. (2003), the use of a water repellent finish on CCA-treated wood significantly reduces the amount of arsenic, as well as copper and chromium, in the run-off water. It was also observed the exposure to UV radiation caused a significant increase in leaching from both finished and unfinished samples of CCA-treated wood. Small amounts of arsenic can be transferred from CCA-treated wood to skin from touching CCA-treated wood surfaces (Hemond and Solo-Gabriele 2004; Kwon et al. 2004; Shalat et al. 2006; Ursitti et al. 2004; Wang et al. 2005).

Breslin and Adler-Ivanbrook (1998) examined the leaching of the copper, chromium, and arsenic from CCA-treated wood in laboratory studies using samples of treated southern yellow pine in solutions simulating estuarine waters. The tank leaching solutions were frequently sampled and replaced to approximate field conditions. Initial 12-hour fluxes ranging from 0.2×10^{-10} to 5.2×10^{-10} mol/mm² d was reported for arsenic. After 90 days, arsenic fluxes decreased to 0.5×10^{-11} – 3.1×10^{-11} mol/mm² d. A study by Cooper (1991) demonstrated that the buffer system used in leaching studies of components from CCA-treated wood can significantly change the amount arsenic released from treated wood. Samples of four

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species of CCA-treated wood were exposed to four acidic leaching solutions. In the samples exposed to water adjusted to pHs of 3.5, 4.5, and 5.5, losses of arsenic after 13 days were generally <7%. However, when a leaching solution of sodium hydroxide and citric acid buffer (pH 5.5) was used, the percent of arsenic leached ranged from 27.4 to 46.7% (Cooper 1991).

Arsenic bioaccumulation depends on various factors, such as environmental setting (marine, estuarine, freshwater), organism type (fish, invertebrate), trophic status within the aquatic food chain, exposure concentrations, and route of uptake (Williams et al. 2006). Bioaccumulation refers to the net accumulation of a chemical by aquatic organisms as a result of uptake from all environmental sources, such as water, food, and sediment, whereas bioconcentration refers to the uptake of a chemical by an aquatic organism through water (EPA 2003b). Biomagnification in aquatic food chains does not appear to be significant (EPA 1979, 1982b, 1983e, 2003b; Mason et al. 2000; Williams et al. 2006).

Bioconcentration of arsenic occurs in aquatic organisms, primarily in algae and lower invertebrates. Both bottom-feeding and predatory fish can accumulate contaminants found in water. Bottom-feeders are readily exposed to the greater quantities of metals, including the metalloid arsenic, which accumulate in sediments. Predators may bioaccumulate metals from the surrounding water or from feeding on other fish, including bottom-feeders, which can result in the biomagnification of the metals in their tissues. An extensive study of the factors affecting bioaccumulation of arsenic in two streams in western Maryland in 1997–1998 found no evidence of biomagnification since arsenic concentrations in organisms tend to decrease with increasing trophic level (Mason et al. 2000). Arsenic is mainly accumulated in the exoskeleton of invertebrates and in the livers of fish. No differences were found in the arsenic levels in different species of fish, which included herbivorous, insectivorous, and carnivorous species. The major bioaccumulation transfer is between water and algae, at the base of the food chain and this has a strong impact on the concentration in fish. National Contaminant Biomonitoring data produced by the Fish and Wildlife Service were used to test whether differences exist between bottom-feeders and predators in tissue levels of metals and other contaminants. No differences were found for arsenic (Kidwell et al. 1995). The bioconcentration factors (BCFs) of bryophytes, invertebrates, and fish (livers) in Swedish lakes and brooks impacted by smelter emissions were 8,700, 1,900–2,200, and 200–800, respectively (Lithner et al. 1995). EPA (2003b) assessed a large dataset of bioaccumulation data for various fish and invertebrate species. BCF values in this dataset ranged from 0.048 to 1,390.

Williams et al. (2006) reviewed 12 studies of arsenic bioaccumulation in freshwater fish, and proposed that BCF and bioaccumulation factor (BAF) values are not constant across arsenic concentrations in

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water. BCF or BAF values from these 12 studies ranged from 0.1 to 3,091. Williams et al. (2006) found that BCF and BAF values appear to be the highest within the range of ambient arsenic concentrations, and decline steeply to relatively low levels as the arsenic concentrations in water increase. Based on this analysis, arsenic concentrations in tissue and BAF values may be a power function of arsenic concentrations in water. EPA (2007b) also reported that for many nonessential metals, including arsenic, accumulation is nonlinear with respect to exposure concentration.

6.3.2 Transformation and Degradation

6.3.2.1 Air

Arsenic is released into the atmosphere primarily as arsenic trioxide or, less frequently, in one of several volatile organic compounds, mainly arsines (EPA 1982b). Trivalent arsenic and methyl arsines in the atmosphere undergo oxidation to the pentavalent state (EPA 1984a), and arsenic in the atmosphere is usually a mixture of the trivalent and pentavalent forms (EPA 1984a; Scudlark and Church 1988). Photolysis is not considered an important fate process for arsenic compounds (EPA 1979).

6.3.2.2 Water

Arsenic in water can undergo a complex series of transformations, including oxidation-reduction reactions, ligand exchange, precipitation, and biotransformation (EPA 1979, 1984a; Sanders et al. 1994; Welch et al. 1988). Rate constants for these various reactions are not readily available, but the factors most strongly influencing fate processes in water include Eh, pH, metal sulfide and sulfide ion concentrations, iron concentrations, temperature, salinity, distribution and composition of the biota, season, and the nature and concentration of natural organic matter (EPA 1979; Farago 1997; Redman et al. 2002; Wakao et al. 1988). Organic arsenical pesticides, such as MSMA, DSMA, and DMA do not degrade by hydrolysis or by aquatic photolysis (EPA 2006). No formation of arsine gas from marine environments has been reported (Tamaki and Frankenberger 1992).

Inorganic species of arsenic are predominant in the aquatic environment. In the pH range of natural waters, the predominant aqueous inorganic As(V) species are the arsenate ions, H_2AsO_4^- and HAsO_4^{2-} ; the predominant inorganic As(III) species is $\text{As}(\text{OH})_3$ (Aurillo et al. 1994; EPA 1982c). As(V) generally dominates in oxidizing environments such as surface water and As(III) dominates under reducing conditions such as may occur in groundwater containing high levels of arsenic. However, the reduction of arsenate to arsenite is slow, so arsenate can be found in reducing environments. Conversely, the oxidation of arsenite in oxidizing environments is moderately slow (half-life, 0.4–7 days in coastal

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systems) and therefore, arsenite can be found in oxidizing environments (Mariner et al. 1996; Sanders et al. 1994). The main organic species in fresh water are MMA and DMA; however, these species are usually present at lower concentrations than inorganic arsenic species (Eisler 1994). (The toxicities of MMA and DMA are discussed in Chapter 3.) Aquatic microorganisms may reduce the arsenate to arsenite, as well as methylate arsenate to its mono- or dimethylated forms (Aurillo et al. 1994; Benson 1989; Braman and Foreback 1973; Edmonds and Francesconi 1987; Sanders et al. 1994). Methylated species are also produced by the biogenic reduction of more complex organoarsenic compounds like arsenocholine or arsenobetaine. Water samples from a number of lakes and estuaries, mostly in California, show measurable concentrations of methylated arsenic (equivalent to 1–59% of total arsenic) (Anderson and Bruland 1991). Within the oxic photic zone, arsenate and DMA were the dominant species. A seasonal study of one lake demonstrated that DMA was the dominant form of arsenic in surface waters during late summer and fall. Methylated species declined and arsenate species increased when the lake turned over in late fall. Mono Lake, a highly alkaline body of water, and four rivers did not have measurable concentrations of methylated arsenic. It was hypothesized that the reason why methylated forms were not detected in Mono Lake was that the extremely high inorganic arsenic concentrations in the lake, 230 μM (17 mg/L), could overwhelm the analysis of small amounts of organic forms. Other possibilities are that the high alkalinity or very high phosphate levels in the water, 260 μM (25 mg/L), are not conducive to biogenic methylation (Anderson and Bruland 1991). Both reduction and methylation of As(V) may lead to increased mobilization of arsenic, since As(III), dimethylarsinates, and monomethylarsonates are much less particle-reactive than As(V) (Aurillo et al. 1994). In the estuarial Patuxet River, Maryland, arsenate concentrations peaked during the summer, at 1.0 $\mu\text{g/L}$ in 1988–1989 (Sanders et al. 1994). In contrast, winter to spring levels were around 0.1 $\mu\text{g/L}$. Arsenite concentrations were irregularly present at low levels during the year. Peaks of DMA occurred at various times, particularly in the winter and late spring and appeared to be linked with algal blooms. The DMA peak declined over several months that was followed by a rise in MMA. The MMA was thought to be occurring as a degradation product of DMA. A similar seasonal pattern of arsenic speciation was observed in Chesapeake Bay. Arsenite methylation took place during the warmer months leading to changes down the main stem of the bay; arsenite production dominated the upper reaches of the bay and methylated species dominated the more saline lower reaches. In coastal waters, reduced and methylated species are present in lower concentrations, around 10–20% of total arsenic (Sanders et al. 1994). In groundwater, arsenic generally exists as the oxyanion of arsenate ($\text{H}_x\text{AsO}_4^{3-x}$) or arsenite ($\text{H}_x\text{AsO}_3^{3-x}$), or both; however, the distribution between arsenite and arsenate is not always predictable based on oxidation-reduction potential (Robertson 1989; Welch et al. 1988).

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6.3.2.3 Sediment and Soil

In soil, arsenic is found as a complex mixture of mineral phases, such as co-precipitated and sorbed species, as well as dissolved species (Roberts et al. 2007). The degree of arsenic solubility in soil will depend on the amount of arsenic distributed between these different mineral phases. The dissolution of arsenic is also affected by particle size. The distribution between these phases may reflect the arsenic source (e.g., pesticide application, wood treatment, tanning, or mining operations), and may change with weathering and associations with iron and manganese oxides and phosphate minerals in the soil (Roberts et al. 2007; Ruby et al. 1999). Davis et al. (1996) reported that in soil in Anaconda, Montana, a smelting site from 1860 to 1980, contained arsenic that is only in a sparingly soluble form, consisting of primarily arsenic oxides and phosphates.

The arsenic cycle in soils is complex, with many biotic and abiotic processes controlling its overall fate and environmental impact. Arsenic in soil exists in various oxidation states and chemical species, depending upon soil pH and oxidation-reduction potential. Under most environmental conditions, inorganic As(V) will exist as a mixture of arsenate anions, H_2AsO_4^- and HAsO_4^{2-} , and inorganic As(III) will exist as H_3AsO_3 . The arsenate and arsenite oxyanions have various degrees of protonation depending upon pH (EPA 1982b; McGeehan 1996). As(V) predominates in aerobic soils, and As(III) predominates in slightly reduced soils (e.g., temporarily flooded) or sediments (EPA 1982b; Sanders et al. 1994). As(III) commonly partitions to the aqueous phase in anoxic environments, and would be more mobile. As(V) usually remains bound to minerals, such as ferrihydrite and alumina, limiting its mobility and bioavailability (Rhine et al. 2006).

Arsenite is moderately unstable in the presence of oxygen; however, it can be found under aerobic conditions as well (Sanders et al. 1994). While arsenate is strongly sorbed by soils under aerobic conditions, it is rapidly desorbed as the system becomes anaerobic. Once it is desorbed, arsenate can be reduced to arsenite, which exhibits greater mobility in soils (McGeehan 1996). Transformations between the various oxidation states and species of arsenic occur as a result of biotic or abiotic processes (Bhumbla and Keefer 1994). While degradation of an organic compound is typically considered complete mineralization, in the case of organic arsenic compounds, the element arsenic itself cannot be degraded. However, the organic portion of the molecule can be metabolized (Woolson 1976).

Arsenicals applied to soils may be methylated by microorganisms to arsines, which are lost through volatilization, and organic forms may be mineralized to inorganic forms. Gao and Burau (1997) reported

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that the overall percentage of DMA and MMA mineralized after 70 days ranged from 3 to 87% in air-dry soil and a soil near saturation, respectively. The rate of demethylation of DMA increased with soil moisture. Over the same 70-day period, arsenic losses as volatile arsines were much lower than mineralization, ranging from 0.001 to 0.4%. Arsine evolution rates followed the order: DMA>MMA>arsenite=arsenate (Gao and Burau 1997). Woolson and Kearney (1973) reported that ^{14}C -labeled DMA degraded differently in soils under aerobic and anaerobic conditions. Under anaerobic conditions, 61% of the applied DMA was converted to a volatile alkyl arsine after 24 weeks, and lost from the soil system. Under aerobic conditions, 35% was converted to a volatile organo-arsenic compound, possibly dimethyl arsine, and 41% was converted to $^{14}\text{CO}_2$ and arsenate after 24 weeks. Similar to microorganisms in soils, Reimer (1989) reported that microorganisms found in natural marine sediments and sediments contaminated with mine-tailings are also capable of methylating arsenic under aerobic and anaerobic conditions. Von Endt et al. (1968) reported that the degradation of ^{14}C -labelled monosodium methanearsonate (MSMA) was found to range from 1.7 to 10% in Dundee silty clay loam soil and Sharkey clay soil after 60 days, respectively. MSMA decomposition to CO_2 was a slow process without a lag period. Sterilized soils were found to produce essentially no $^{14}\text{CO}_2$ (0.7%) after 60 days, indicating that soil bacteria contributed to the decomposition of MSMA (Von Endt et al. 1968). Akkari et al. (1986) studied the degradation of MSMA in various soils. At 20% water content, half-lives of 144, 88, and 178 days were reported in Sharkey clay, Taloka silt loam, and Steele-Crevasse sand loam, respectively. The Sharkey soil with the highest clay content was expected to have the greatest adsorptive capacity for both water and MSMA, reducing the amount of MSMA available in the soil solution to microorganisms that degrade the MSMA. The half-lives were 25, 41, and 178 days under anaerobic (flooded) conditions in Sharkey clay, Taloka silt loam, and Steele-Crevasse sand loam, respectively. Under flooded conditions, MSMA degradation occurs by reductive methylation to form arsinite and alkylarsine gas. The authors attributed the longer half-lives for MSMA degradation in the Steele-Crevasse sand loam soil to its low organic matter content, which may have supported fewer microbial populations needed for oxidation demethylation under aerobic conditions. Under flooded conditions, anaerobiosis is expected to be slowest in low organic matter sandy loam soils (Akkari et al. 1986).

Organic arsenical pesticides, such as MSMA, DSMA, and DMA, do not degrade by hydrolysis or by soil photolysis (EPA 2006).

Roxarsone (3-nitro-4-hydroxyphenylarsonic acid) used in poultry feed is found excreted unchanged in poultry litter (bedding and manure). Roxarsone found in poultry litter, which is used to amend agricultural soil, was found to degrade to arsenate in approximately 3–4 weeks upon composting

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(Garbarino et al. 2003). In addition, the arsenic in poultry litter was found to be easily mobilized by water; however, its leach rate from amended soils was slow enough that it accumulated in soils (Rutherford et al. 2003).

A sequential fractionation scheme was used to assess the chemical nature, and thus the potential bioavailability, of arsenic at cattle dip sites in Australia where sodium arsenite was used extensively in cattle dips from the turn of the century until the early 1950s (McLaren et al. 1998). Most sites contained substantial amounts, 13% on the average, of arsenic in the two most labile fractions indicating a high potential for bioaccessibility and leaching. The bulk of the arsenic appeared to be associated with amorphous iron and aluminum minerals in soil. Similarly, arsenic in soil and mine waste in the Tamar Valley in England was found to be concentrated in a fraction associated with iron and organic-iron (Kavanagh et al. 1997). Laboratory studies were performed to assess the phase partitioning of trace metals, including the metalloid arsenic, to sediment from the Coeur d'Alene River, a mining area of Idaho, and the release of metals under simulated minor and major flooding events (LaForce et al. 1998). Arsenic was primarily associated with the iron and manganese oxides as seen by its large release when these oxides were reduced. Arsenic levels were comparatively low in the organic fraction and remaining residual fraction and negligible in the extractible fractions.

6.3.2.4 Other Media

Carbonell-Barrachina et al. (2000) found the speciation and solubility of arsenic in sewage sludge suspensions to be affected by pH and Eh. Under oxidizing conditions, the solubility of arsenic was low, with a major portion of the soluble arsenic present as organic arsenic compounds, mainly dimethylarsinic acid (approximately 74% of the total arsenic in solution). Under moderately reducing conditions (0–100 mV), inorganic arsenic accounted for the majority (90%) of the total arsenic in solution, and the solubility of arsenic was increased due to dissolution of iron oxyhydroxides. Under strongly reducing conditions (-250 mV), arsenic solubility was decreased by the formation of insoluble sulfides. The pH of the solution was also found to influence the speciation and solubility of arsenic. At neutral pH, the solubility of arsenic was at its maximum, and decreased under acidic or alkaline conditions. Inorganic arsenic species were the dominant species at pH 5.0; at pH 6.5, the major soluble forms were organic arsenic species. The biomethylation of arsenic was limited at acidic pH, and was at its maximum at near neutral pH (Carbonell-Barrachina et al. 2000).

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6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to arsenic depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of arsenic 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 arsenic 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. The analytical methods available for monitoring arsenic in a variety of environmental media are detailed in Chapter 7.

6.4.1 Air

Arsenic in ambient air is usually a mixture of particulate arsenite and arsenate; organic species are of negligible importance except in areas of substantial methylated arsenic pesticide application or biotic activity (EPA 1984a). Mean levels in ambient air in the United States have been reported to range from <1 to 3 ng/m^3 in remote areas and from 20 to 30 ng/m^3 in urban areas (Davidson et al. 1985; EPA 1982c; IARC 1980; NAS 1977a). EPA conducted a modeling study with the Assessment System for Population Exposure Nationwide (ASPEN) in which estimates of emissions of hazardous air pollutants were used to estimate ambient concentrations (Rosenbaum et al. 1999). Using 1990 data to estimate total emissions of arsenic in the conterminous 48 states, excluding road dust or windblown dust from construction or agricultural tilling, the 25th percentile, median, and 75th percentile arsenic concentration were estimated to be 9 , 20 , and 30 ng/m^3 , respectively. Maps illustrating the amount of toxic air pollutant emissions, including arsenic compounds, by county in 1996 for the 48 coterminous states of the United States as well as Puerto Rico and the Virgin Islands are available on the internet at <http://www.epa.gov/ttn/atw/-nata/mapemis.html>, as of March 2005. Schroeder et al. (1987) listed ranges of arsenic concentrations in air of 0.007 – 1.9 , 1.0 – 28 , and 2 – $2,320 \text{ ng/m}^3$ in remote, rural, and urban areas, respectively. The average annual arsenic concentration in air at Nahant, Massachusetts, just north of Boston, between September 1992 and September 1993, was 1.2 ng/m^3 ; 75% of the arsenic was associated with fine ($<2.5 \mu\text{m}$) particles. The long-term means of the ambient concentrations of arsenic measured in urban areas of the Great Lakes region from 1982 to 1993 ranged from 4.2 to 9.6 ng/m^3 (Pirrone and Keeler 1996). Large cities generally have higher arsenic air concentrations than smaller ones due to emissions from coal-fired power plants (IARC 1980), but maximum 24-hour concentrations generally are $<100 \text{ ng/m}^3$ (EPA 1984a). In the spring of 1990, aerosols and cloud water that were sampled by aircraft at an altitude of 1.2 – 3 km above the Midwestern United States had a mean mixed layer arsenic concentration of $1.6 \pm 0.9 \text{ ng/m}^3$ (Burkhard et al. 1994). A mean arsenic concentration of $1.0 \pm 0.5 \text{ ng/m}^3$ was reported at

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Mayville, New York, a site 400 km to the northwest of the sampling area and directly downwind on most days.

Arsenic was monitored at an application site in the San Joaquin Valley, California and at four sites in nearby communities in 1987 where sodium arsenite was used as a fungicide on tokay grapes (Baker et al. 1996). The maximum arsenic concentration measured 15–20 meters from the edge of the field was 260 ng/m³. The maximum arsenic concentration at four community sites in the area was 76 ng/m³. The concentration at an urban background site was 3 ng/m³ (Baker et al. 1996). Sodium arsenite is no longer registered in California (Baker et al. 1996). The highest historic arsenic levels detected in the atmosphere were near nonferrous metal smelters, with reported concentrations up to 2,500 ng/m³ (IARC 1980; NAS 1977a; Schroeder et al. 1987).

Arsenic air concentrations measured in several indoor public places (e.g., cafeteria, coffee house, music club, Amtrak train, and several restaurants) with environmental tobacco smoke (ETS) ranged from <0.1 to 1 ng/m³, with a mean of 0.4±0.3 ng/m³. Sites that were ETS-free (university office and library) had arsenic concentrations <0.13 ng/m³ (Landsberger and Wu 1995). The Toxic Exposure Assessment at Columbia/Harvard (TEACH) study measured levels of various toxics in New York City air in 1999. Exposures were assessed in a group of 46 high school students in West Central Harlem. Mean arsenic concentrations in summer home outdoor, home indoor, and personal air of the participants were 0.37, 0.40, and 0.45 ng/m³, respectively (Kinney et al. 2002). Detected arsenic concentrations in indoor and outdoor air collected as part of the National Human Exposure Assessment Survey (NHEXAS) in Arizona ranged from 3.4 to 22.3 and from 3.5 to 25.7 ng/m³, respectively, with 71 and 68% below the detection limit (1.8–14.3 ng/m³) (O'Rourke et al. 1999).

6.4.2 Water

Arsenic is widely distributed in surface water, groundwater, and finished drinking water in the United States. A survey of 293 stations in two nationwide sampling networks on major U.S. rivers found median arsenic levels to be 1 µg/L; the 75th percentile level was 3 µg/L (Smith et al. 1987). Arsenic was detected in 1,298 of 3,452 surface water samples recorded in the STORET database for 2004 at concentrations ranging from 0.138 to 1,700 µg/L in samples where arsenic was detected (EPA 2005c). Two streams in western Maryland that were the focus of a major bioaccumulation study in 1997–1998 had arsenic concentrations of 0.370±0.200 and 0.670±0.460 µg/L (Mason et al. 2000). Surface water will be impacted by runoff from polluted sites. An average arsenic concentration of 5.12 µg/L was reported in

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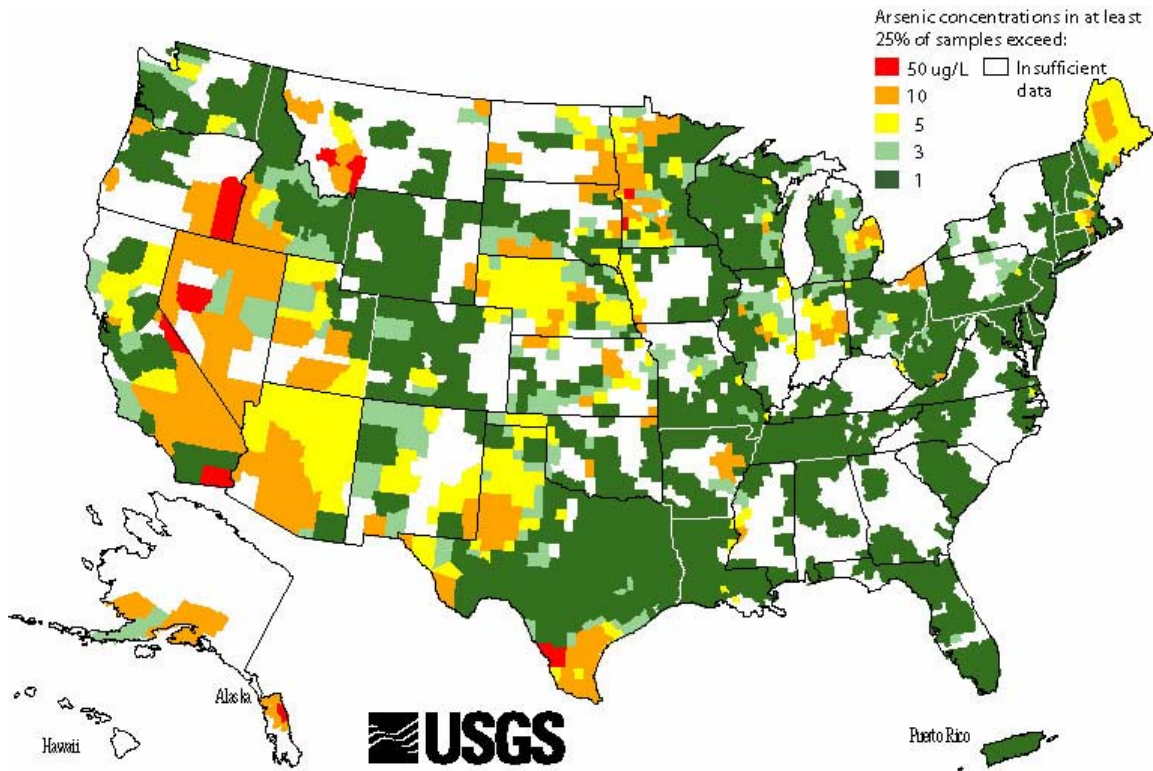
water from Moon Lake, a Mississippi River alluvial floodplain in northwest Mississippi. Intensive cultivation has occurred in this area, including cotton, soybeans, and rice (Cooper and Gillespie 2001). Hard-rock mining activities occurred in the southern part of Colorado and New Mexico north of Taos since the latter part of the 19th century until recently, which have impacted the Rio Grande and its tributaries. A mean arsenic concentration of approximately 0.8 µg/L was reported for the main stem of the Rio Grande sampled in June and September 1994. Arsenic concentrations in the Alamosa River, Colorado were 0.11 and 0.14 µg/L in June and September 1994, respectively, and 1.4 µg/L in Big Arsenic Spring, New Mexico in September 1994 (Taylor et al. 2001). Arsenic concentrations in water from watersheds in Black Hills, South Dakota, an area impacted by gold mining activities ranged from 2.5 to 55 µg/L and from 1.7 to 51 µg/L in unfiltered and filtered samples, respectively; concentrations from reference areas ranged from 1.1 to 3.4 µg/L and from 0.9 to 1.9 µg/L in unfiltered and filtered samples, respectively (May et al. 2001). Arsenic concentrations ranged from 0.29 to 34.0 µg/L in water samples from Wakulla River and St. Joseph Bay North, along the Florida Panhandle; arsenic contamination in this area is likely to result from nonpoint source pollution (Philp et al. 2003).

Data on total arsenic in surface water from a number of seas and oceans show levels of <1 µg/L, except in the Antarctic Ocean and Southwest Pacific Oceans where the levels are 1.1 and 1.2 µg/L, respectively. Levels in coastal waters and estuaries are generally somewhat higher, in the range of 1–3 µg/L. However, estuarine water in Salinas, California had arsenic levels of 7.42 µg/L (Francesconi et al. 1994). The dissolved arsenic concentration in water at 40 sites in the Indian River Lagoon System in Florida ranged from 0.35 to 1.6 µg/L with a mean of 0.89±0.34 µg/L (Trocine and Trefry 1996). Thermal waters generally have arsenic levels of 20–3,800 µg/L, although levels as high as 276,000 µg/L have been recorded (Eisler 1994).

Arsenic levels in groundwater average about 1–2 µg/L, except in some western states with volcanic rock and sulfidic mineral deposits high in arsenic, where arsenic levels up to 3,400 µg/L have been observed (IARC 1980; Page 1981; Robertson 1989; Welch et al. 1988). In western mining areas, groundwater arsenic concentrations up to 48,000 µg/L have been reported (Welch et al. 1988). Arsenic concentrations in groundwater samples collected from 73 wells in 10 counties in southeast Michigan in 1997 ranged from 0.5 to 278 µg/L, with an average of 29 µg/L. Most (53–98%) of the arsenic was detected as arsenite (Kim et al. 2002). The U.S. Geological Survey mapped concentrations of arsenic in approximately 31,350 groundwater samples collected between 1973 and 2001; the counties in which at least 25% of wells exceed various levels are shown in Figure 6-2 (USGS 2007a). Most arsenic in natural waters is a

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Figure 6-2. Counties in Which at Least 25% of Wells Exceed Different Arsenic Levels



Source: USGS 2007a

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mixture of arsenate and arsenite, with arsenate usually predominating (Braman and Foreback 1973; EPA 1982c, 1984a). Methylated forms have also been detected in both surface water and groundwater, at levels ranging from 0.01 to 7.4 µg/L (Braman and Foreback 1973; Hood 1985), with most values below 0.3 µg/L (Hood 1985). In a survey of shallow groundwater quality in the alluvial aquifer beneath a major urban center, Denver, Colorado, arsenic levels in the 30 randomly-chosen wells sampled had median levels of <1 µg/L; the maximum level was 33 µg/L (Bruce and McMahon 1996). Arsenic levels in groundwater sometimes exceeded the EPA maximum contaminant level (MCL), which was 50 µg/L at the time, in the Willamette Valley, Oregon and a nine-county region of southeastern Michigan (USGS 1999b, 1999c).

Arsenic has also been detected in rainwater at average concentrations of 0.2–0.5 µg/L (Welch et al. 1988). This range is consistent with that found in a 1997–1998 study in western Maryland, which was the focus of a major bioaccumulation study (Mason et al. 2000). Arsenic levels in wet deposition in the watershed as well as throughfall into the two streams were 0.345 ± 0.392 , 0.400 ± 0.400 , and 0.330 ± 0.250 µg/L, respectively. Median arsenic concentrations in 30-day rainwater composite samples collected May–September 1994 from eight arctic catchments in northern Europe at varying distances and wind directions from the emissions of a Russian nickel ore mining, roasting, and smelting industry on the Kola Peninsula ranged from 0.07 to 12.3 µg/L (Reimann et al. 1997). Rain and snow samples were collected during the fall of 1996 and winter of 1997 at eight locations in a semi-circular pattern radiating out (2–15 km) in the direction of the prevailing wind from the Claremont incinerator located in New Hampshire. This incinerator processes 200 tons of solid waste per day. Arsenic concentrations in rainwater and snow ranged from 0.020 to 0.079 µg/L and from 0.80 to 1.28 µg/L, respectively (Feng et al. 2000).

Drinking water is one of the most important sources of arsenic exposure. Surveys of drinking water in the United States have found that >99% of public water supplies have arsenic concentrations below the EPA MCL, which was 50 µg/L at the time (EPA 1984a). In an EPA study of tap water from 3,834 U.S. residences, the average value was 2.4 µg/L (EPA 1982c).

Before the MCL for arsenic in drinking water was lowered from 50 to 10 µg/L, studies were undertaken to ascertain how different standards would affect compliance. One such survey sponsored by the Water Industry Technical Action Fund was the National Arsenic Occurrence Survey (NAOS). NAOS was based on a representational survey of public water systems defined by source type, system size, and geographical location. Additionally, it included a natural occurrence factor, a stratifying variable that could qualitatively describe the likelihood of arsenic occurrence in the supply. To predict finished water

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arsenic concentrations, data on the water treatment options, efficiency, and frequency of use were factored in. The results of the NAOS are presented in Table 6-3. The NAOS results are in general agreement with two older and more limited national surveys, EPA's National Inorganics and Radionuclides Survey (NIRS) and the Metropolitan Water District of Southern California Survey (MWDSC). The percentages of water systems that would be out of compliance are estimated to be 1.7, 3.6, 9.3, and 20.7% for arsenic MCLs of 20, 10, 5, and 2 µg/L, respectively. Arsenic concentrations were determined in drinking in EPA Region V (Indiana, Illinois, Michigan, Minnesota, Ohio, and Wisconsin) as part of the NHEXAS; mean arsenic concentration in flushed and standing tap water were both 1.1 µg/L (Thomas et al. 1999). A review by Frost et al. (2003) of existing data from the EPA Arsenic Occurrence and Exposure Database, as well as additional data from state health and environmental departments and water utilities found that 33 counties in 11 states had estimated mean drinking water arsenic concentrations of 10 µg/L or greater. Eleven counties had mean arsenic concentrations of ≥ 20 µg/L, and two counties had mean arsenic concentrations of ≥ 50 µg/L (Frost et al. 2003).

The north central region and the western region of the United States have the highest arsenic levels in surface water and groundwater sources, respectively. In a study of drinking water from New Hampshire, arsenic concentrations ranged from <0.01 to 180 µg/L in the 793 households tested. More than 10% of the private wells had arsenic concentrations >10 µg/L, and 2.5% had levels >50 µg/L (Karagas et al. 1998). In New Hampshire, 992 randomly selected household water samples were analyzed for arsenic levels and the results for domestic well users were compared with those for users of municipal water supplies (Peters et al. 1999). The concentrations ranged from <0.0003 to 180 µg/L, with water from domestic wells containing significantly more arsenic than water from municipal supplies; the median concentration of the former was about 0.5 µg/L and the latter was 0.2 µg/L. None of the municipal supplies exceeded an arsenic concentration of 50 µg/L, and 2% of the domestic wells were found to have arsenic concentrations that exceeded 50 µg/L. Approximately 2% of the municipal water users have water with arsenic levels exceeding 10 µg/L compared with 13% of domestic wells. Twenty-five percent of domestic wells and 5% of municipal supplies were found to have arsenic concentrations exceeding 2 µg/L. The highest arsenic levels in New Hampshire are associated with bedrock wells in the south eastern and south central part of the state (Peters et al. 1999). In a study of arsenic in well water supplies in Saskatchewan, Canada, 13% of samples were >20 µg/L and one sample exceeded 100 µg/L (Thompson et al. 1999). It was noted that the samples with high arsenic levels were derived from sites that were in near proximity to each other, indicating the presence of 'hot spots' with similar geological characteristics. As part of an epidemiological study, Engel and Smith (1994) investigated the levels of arsenic in drinking

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Table 6-3. Regional Occurrence of Arsenic in U.S. Water Sources and Finished Drinking Water

Geographical region	Arsenic concentration in µg/L			
	<1	1–5	5–20	>20
<i>Occurrence in U.S. surface water sources</i>				
Region 1. New England	50	50	0	0
Region 2. Mid-Atlantic	84	12	4	0
Region 3. South East	93	7	0	0
Region 4. Midwest	24	76	0	0
Region 5. South Central	32	55	13	0
Region 6. North Central	33	22	33	0
Region 7. Western	42	58	0	0
<i>Occurrence in U.S. groundwater sources</i>				
Region 1. New England	71	21	7	0
Region 2. Mid-Atlantic	81	4	11	4
Region 3. South East	82	14	2	0
Region 4. Midwest	40	40	15	5
Region 5. South Central	68	27	15	0
Region 6. North Central	30	40	30	0
Region 7. Western	24	34	28	14
<i>Occurrence in U.S. finished surface water supplies</i>				
Region 1. New England	88	12	0	0
Region 2. Mid-Atlantic	92	8	0	0
Region 3. South East	100	0	0	0
Region 4. Midwest	73	27	0	0
Region 5. South Central	74	19	7	0
Region 6. North Central	44	44	0	12
Region 7. Western	42	58	0	0
<i>Occurrence in U.S. finished groundwater supplies</i>				
Region 1. New England	79	21	0	0
Region 2. Mid-Atlantic	81	4	11	4
Region 3. South East	94	4	2	0
Region 4. Midwest	58	27	12	3
Region 5. South Central	61	27	12	0
Region 6. North Central	40	50	10	0
Region 7. Western	20	40	22	12

Source: National Arsenic Occurrence Survey (Frey and Edwards 1997)

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water throughout the United States between 1968 and 1984. They found that 30 counties in 11 states had mean arsenic levels of $>5 \mu\text{g/L}$, with a range of $5.4\text{--}91.5 \mu\text{g/L}$; 15 counties had mean levels from 5 to $10 \mu\text{g/L}$; 10 counties had mean levels from 10 to $20 \mu\text{g/L}$; and 5 counties had levels $>20 \mu\text{g/L}$. The highest levels were found in Churchill County, Nevada, where 89% of the population was exposed to a mean arsenic concentration of $100 \mu\text{g/L}$ and 11% to a mean of $27 \mu\text{g/L}$. A study by Frost et al. (2003) identified 33 counties from 11 states in which the average arsenic concentration of at least 75% of public wells was $>10 \mu\text{g/L}$. Arsenic concentrations in drinking water from these counties ranged from 10.3 to $90.0 \mu\text{g/L}$ in Pinal, Arizona and Churchill, Nevada, respectively (Frost et al. 2003).

Many communities have high levels of arsenic in their drinking water because of contamination or as a result of the geology of the area. In Millard County, Utah, seven towns had median and maximum arsenic levels of $18.1\text{--}190.7$ and $125\text{--}620 \mu\text{g/L}$, respectively, in their drinking water (Lewis et al. 1999). The mean arsenic concentration in tap water from homes in Ajo, Arizona, about 2 miles from an open pit copper mine and smelter was $90 \mu\text{g/L}$ (Morse et al. 1979). The town's water was supplied from five deep wells.

Countries such as Mexico, Bangladesh, India, Chile, Argentina, and Vietnam have highly elevated levels of arsenic in drinking water in some regions (Bagla and Kaiser 1996; Berg et al. 2001; Tondel et al. 1999; WHO 2001; Wyatt et al. 1998a, 1998b). In Bangladesh and West Bengal, the soil naturally contains high levels of arsenic, which leaches into the shallow groundwater that is tapped for drinking water. In West Bengal, India, it is estimated that more than one million Indians are drinking arsenic-laced water and tens of millions more could be at risk in areas that have not been tested for contamination. Analysis of 20,000 tube-well waters revealed that 62% have arsenic at levels above the World Health Organization (WHO) permissible exposure limit (PEL) in drinking water of $10 \mu\text{g/L}$, with some as high as $3,700 \mu\text{g/L}$ (Bagla and Kaiser 1996). Analysis of 10,991 and 58,166 groundwater samples from 42 and 9 arsenic-affected districts in Bangladesh and West Bengal were found to have arsenic levels that were 59 and 34%, respectively, above $50 \mu\text{g/L}$ (Chowdhury et al. 2000). Berg et al. (2001), studied the arsenic contamination of the Red River alluvial tract in Hanoi, Vietnam and the surrounding rural areas. Arsenic concentrations in groundwater from private small-scale tube-wells averaged $159 \mu\text{g/L}$, ranging from 1 to $3,050 \mu\text{g/L}$. Arsenic concentrations ranged from 37 to $320 \mu\text{g/L}$ in raw groundwater pumped from the lower aquifer for the Hanoi water supply (Berg et al. 2001). Several investigators have noticed a correlation between high levels of arsenic and fluoride in drinking water (Wyatt et al. 1998a, 1998b). Arsenic concentrations in drinking water from four villages in Bangladesh ranged from 10 to $2,040 \mu\text{g/L}$ (Tondel et al. 1999).

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6.4.3 Sediment and Soil

Arsenic is widely distributed in the Earth's crust, which contains about 3.4 ppm arsenic (Wedepohl 1991). It is mostly found in nature minerals, such as realgar (As_4S_4), orpiment (As_2S_3), and arsenolite (As_2O_3), and only found in its elemental form to a small extent. There are over 150 arsenic-bearing minerals (Budavari et al. 2001; Carapella 1992). Arsenic concentrations in soils from various countries can range from 0.1 to 50 $\mu\text{g/g}$ and can vary widely among geographic regions. Typical arsenic concentrations for uncontaminated soils range from 1 to 40 $\mu\text{g/g}$, with the lowest concentrations in sandy soils and soils derived from granites. Higher arsenic concentrations are found in alluvial soils and soils with high organic content (Mandal and Suzuki 2002). Arsenic in soil may originate from the parent materials that form the soil, industrial wastes, or use of arsenical pesticides. Geological processes that may lead to high arsenic concentrations in rock and subsequently the surrounding soil include hydrothermic activity and pegmatite formation (Peters et al. 1999). In the first case, thermal activity results in the dissolution and transport of metals, including the metalloid arsenic, which are precipitated in fractures in rocks. In the second process, cooling magmas may concentrate metals that are injected into rocks, crystallizing as pegmatites. Areas of volcanic activity include large areas of California, Hawaii, Alaska, Iceland, and New Zealand.

The U.S. Geological Survey reports the mean and range of arsenic in soil and other surficial materials as 7.2 and <0.1–97 $\mu\text{g/g}$, respectively (USGS 1984). The concentrations of arsenic in 445 Florida surface soils ranged from 0.01 to 50.6 $\mu\text{g/g}$ (Chen et al. 1999). The median, arithmetic mean, and geometric mean were 0.35, 1.34 ± 3.77 , and 0.42 ± 4.10 $\mu\text{g/g}$, respectively. Chirenje et al. (2003b) reported a geometric mean arsenic concentrations of 0.40 (0.21–660) and 2.81 (0.32–110) $\mu\text{g/g}$ in surface soil samples (0–20 cm) collected in May–June 2000 from Gainesville and Miami, Florida, respectively. The geometric mean arsenic concentration in 50 California soils was 2.8 $\mu\text{g/g}$ (Chen et al. 1999). In the Florida surface soils, arsenic was highly correlated ($\alpha=0.0001$) with the soil content of clay, organic carbon, CEC, total iron, and total aluminum. Arsenic tends to be associated with clay fractions and iron and manganese oxyhydroxides. Soils of granitic origin are generally low in arsenic, about 4 $\mu\text{g/g}$, whereas arsenic in soils derived from sedimentary rocks may be as high as 20–30 $\mu\text{g/g}$ (Yan-Chu 1994). Soils overlying arsenic-rich geologic deposits, such as sulfide ores, may have soil concentrations two orders of magnitude higher (NAS 1977a).

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Soils in mining areas or near smelters may contain high levels of arsenic. Arsenic concentrations up to 27,000 $\mu\text{g/g}$ were reported in soils contaminated with mine or smelter wastes (EPA 1982b). Soils at an abandoned mining site in the Tamar Valley in southwest England have arsenic concentrations that may exceed 50,000 $\mu\text{g/g}$ (Erry et al. 1999). The average arsenic levels in the top 2 cm of different soil types in the vicinity of a former copper smelter in Anaconda, Montana, ranged from 121 to 236 $\mu\text{g/g}$; levels were significantly related to proximity and wind direction to the smelter site (Hwang et al. 1997a). Smelter fallout can contaminate land miles from the source. Soils in mainland southern King County were studied for the presence of arsenic and lead (WSDOE 2005). Soil samples were collected in the fall of 1999 and the spring of 2001 from locations around the ASARCO smelter, which operated in Ruston from the 1890s to 1986. The study area ran roughly from the I-90 corridor south to the King-Pierce county line, from the Puget Sound shore to the Cascade foothills. Almost all of the contamination was found was in the 0–6-inch depths of the cores samples; 62 of the 75 samples were found to have arsenic levels above 20 ppm (WSDOE 2005).

Soil on agricultural lands treated with arsenical pesticides may retain substantial amounts of arsenic. One study reported an arsenic concentration of 22 $\mu\text{g/g}$ in treated soil compared to 2 $\mu\text{g/g}$ for nearby untreated soil (EPA 1982b). Arsenic was measured in soil samples taken from 10 potato fields in Suffolk County on Long Island, New York, where sodium arsenite had been used for vine control and fall weed control for many years. Lead arsenate also may have been used as an insecticide in certain areas. The mean arsenic levels taken at a depth of 0–18 cm from each of the 10 fields ranged from 27.8 ± 5.44 $\mu\text{g/g}$ dry weight ($n=10$) to 51.0 ± 7.40 $\mu\text{g/g}$ dry weight ($n=10$). These levels were markedly higher than the level of 2.26 ± 0.33 $\mu\text{g/g}$ ($n=10$) for untreated control soils (Sanok et al. 1995). A survey was conducted in 1993 to determine the concentrations of arsenic and lead in soil samples from 13 old orchards in New York State. Lead arsenate was used for pest control in fruit orchards for many years, mainly from the 1930s to 1960s, and residues remain in the soil. Concentrations of arsenic ranged from 1.60 to 141 $\mu\text{g/g}$ dry weight (Merwin et al. 1994). Arsenic and lead concentrations were also measured in former orchard soils contaminated by lead arsenate from the Hanford site in Washington State. The mean arsenic concentration in surface (5–10 cm) and subsurface (10–50 cm) soils were 30 (2.9–270) and 74 (32–180) $\mu\text{g/g}$ dry weight, respectively (Yokel and Delistraty 2003). Average arsenic concentration of 5.728, 5.614, and 6.746 $\mu\text{g/g}$ were reported in soils, lake sediments, and wetland sediments, respectively, from Moon Lake, a Mississippi River alluvial floodplain in northwest Mississippi. Intensive cultivation has occurred in this area, including cotton, soybeans, and rice (Cooper and Gillespie 2001). A geometric mean arsenic concentration of 20.6 mg/kg (range 4.6–340 mg/kg) was reported soil collected during the

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summer and fall of 2003 from 85 homes in Middleport, New York, where historical pesticide manufacturing was associated with arsenic in the soil (Tsuji et al. 2005).

The Washington State Area-wide Soil Contamination Project provides various data on arsenic contamination in soils across Washington State (Washington State 2006). Arsenic concentrations within areas affected by area-wide soil contamination are highly variable, ranging from natural background levels to >3,000 ppm in smelter areas. Generally, average arsenic concentrations in soil at developed properties are <100 ppm. Areas affected by smelter emissions in King, Pierce, Snohomish, and Stevens Counties have a higher likelihood of arsenic soil contamination than other areas of the State due to historical emissions from metal smelters located in Tacoma, Harbor Island, Everett, Northport, and Trail, British Columbia. Areas where apples and pears were historically grown, such as Chelan, Spokane, Yakima, and Okanogan Counties, also have a higher likelihood of arsenic soil contamination than other areas due to the past use of lead arsenate pesticides. Generally, arsenic contamination in soils from historical smelter emissions and historical use of lead-arsenate pesticides is found in the upper 6–18 inches of soil (Washington State 2006).

The New Jersey Department of Environmental Protection (Historic Pesticide Contamination Task Force 1999) reported on the analysis of soil samples collected from 18 sites for various pesticide residues, including arsenic, from current and former agricultural sites in New Jersey in order to assess contamination from historic pesticide use. Arsenic was detected in all 463 samples, with concentrations ranging from 1.4 to 310 ppm.

Natural concentrations of arsenic in sediments are usually <10 µg/g dry weight, but can vary widely around the world (Mandal and Suzuki 2002). Sediment arsenic concentrations reported for U.S. rivers, lakes, and streams range from about 0.1 to 4,000 µg/g (Eisler 1994; Heit et al. 1984; NAS 1977a; Welch et al. 1988). During August through November 1992 and August 1993, bed sediment in the South Platte River Basin (Colorado, Nebraska, and Wyoming) was sampled and analyzed for 45 elements, including arsenic. The range of arsenic found was 2.8–31 µg/g dry weight and the geometric mean (n=23) was 5.7 µg/g (Heiny and Tate 1997). The arsenic concentration in surface sediment (0–2 cm) at 43 sites in the Indian River Lagoon System in Florida ranged from 0.6 to 15 µg/g dry weight with a mean of 5.0±3.9 µg/g (Trocine and Trefry 1996). Arsenic levels were well correlated with those of aluminum. Correlation with aluminum levels is used to normalize sediment level concentrations to natural levels in Florida estuaries. Surficial sediments collected from 18 locations in 3 major tributaries to Newark Bay, New Jersey, were analyzed for 7 toxic metals, including arsenic (Bonnievie et al. 1994). The highest

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concentrations of arsenic were found in the Rahway River adjacent to a chemical plant, 58 $\mu\text{g/g}$ dry weight, and in the Hackensack River adjacent to a coal-fired power plant, 49 $\mu\text{g/g}$. The average arsenic concentration for all sediments was 17 ± 16 $\mu\text{g/g}$. Sediments collected from seven sites in Baltimore Harbor, Maryland, at five seasonal periods between June 1987 and June 1988 had a geometric mean maximum of 7.29 $\mu\text{g/g}$ dry weight and a geometric mean minimum of 1.25 $\mu\text{g/g}$ (Miles and Tome 1997). This harbor is one of two sub-tributaries of the Chesapeake Bay where contaminants have been discharged on a large scale.

The upper Clark Fork River basin in western Montana is widely contaminated by metals from past mining, milling, and smelting activities. In a 1991 study, arsenic levels were determined in sediment along the river and in a reservoir 205 km downstream. Total arsenic in sediments from Clark Fork River decreased from 404 $\mu\text{g/g}$ dry weight at the farthest upstream sampling station to 11 $\mu\text{g/g}$, 201 km downstream. Sediment samples from the Milltown Reservoir had arsenic concentrations ranging from 6 to 56 $\mu\text{g/g}$ (Brumbaugh et al. 1994). Total recoverable arsenic in nonfiltered pore water from the Clark Fork River decreased from 1,740 $\mu\text{g/L}$ at the farthest upstream sampling station to 31 $\mu\text{g/L}$ at the 201 km station (Brumbaugh et al. 1994). The Coeur d'Alene river basin in northern Idaho has been contaminated with heavy metals from mining and smelting operations since 1885 (Farag et al. 1998). A 1994 study determined the metal content of sediment, biofilm, and invertebrates at 13 sites in the basin, 10 with historic mining activity and 3 reference sites. The mean arsenic levels in sediment at the mining sites ranged from 8.3 to 179.0 $\mu\text{g/g}$ dry weight, compared to 2.4–13.1 $\mu\text{g/g}$ dry weight at the reference sites. The mean arsenic levels in biofilm adhering to rock in the water at the mining sites ranged from 7.5 to 155.8 $\mu\text{g/g}$ dry weight, compared to 7.2–27.3 $\mu\text{g/g}$ dry weight at the reference sites. In Whitewood Creek, South Dakota, where as much as 100 million tons of mining and milling waste derived from gold mining activities were discharged between 1876 and 1977, mean and maximum sediment arsenic concentrations were 1,920 and 11,000 $\mu\text{g/g}$, respectively (USGS 1987). Uncontaminated sediment had mean arsenic levels of 9.2 $\mu\text{g/g}$. Arsenic concentrations in surface (0–5 cm) sediments from watersheds in Black Hills, South Dakota, an area impacted by gold mining activities, ranged from 23 to 1,951 $\mu\text{g/g}$ dry weight; concentrations from reference areas ranged from 10 to 58 $\mu\text{g/g}$ dry weight (May et al. 2001). Swan Lake, a sub-bay of Galveston Bay in Texas is a highly industrial area that received runoff from a tin smelter in the 1940s and 1950s. Surface sediments at 17 sites where oysters and mussels were collected ranged from 4.53 to 103 $\mu\text{g/g}$ (Park and Presley 1997). A site in the channel leading from the old smelter had arsenic levels of 568 $\mu\text{g/g}$. Surface sediment was less contaminated than deeper sediment, indicating less arsenic input recently than in the past as a result of the smelter closing.

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It has been suggested that the wood preservative most commonly used in dock pilings and bulkheads, CCA, can be toxic to estuarine organisms. Wendt et al. (1996) measured arsenic in surface sediments and oysters from creeks with high densities of docks and from nearby reference creeks with no docks. The average concentrations in the sediments ranged from 14 to 17 $\mu\text{g/g}$ throughout the study area, which is within the range of natural background levels. Weis et al. (1998) sampled sediments along a 10-m transect from CCA-treated wood bulkheads from four Atlantic coast estuaries. Arsenic concentrations were highest in the fine-grained portion of the sediments near the CCA-treated bulkhead (0–1 m); arsenic concentrations were generally at reference levels at distances >1 m from the bulkheads (Weis et al. 1998).

Soils below and around play structures constructed from CCA-treated wood in the City of Toronto, Canada were sampled and analyzed for inorganic arsenic (Ursitti et al. 2004). A mean arsenic concentration of 2.1 $\mu\text{g/g}$ (range 0.5–10 $\mu\text{g/g}$) was reported in soil samples taken within 1 m of the CCA-treated wood for all play structures. Soil samples that were collected 10 m from the play structures served as a background had arsenic concentration of 2.4 $\mu\text{g/g}$ (range 0.5–13 $\mu\text{g/g}$). A mean arsenic concentration of 6.2 $\mu\text{g/g}$ (range 0.5–47.5 $\mu\text{g/g}$) was reported in soil samples taken below CCA-treated wood for all play structures. Of the 217 play structures in the study, 32 had arsenic concentrations under the play structures that exceeded the Canadian federal soil guidelines with arsenic concentrations ranging from 12.4 to 47.5 $\mu\text{g/g}$. From this study, the authors concluded that arsenic does not migrate laterally, but does accumulate in soil under elevated platforms constructed from CCA-treated wood (Ursitti et al. 2004).

6.4.4 Other Environmental Media

Low levels of arsenic are commonly found in food; the highest levels are found in seafood, meats, and grains. Typical U.S. dietary levels of arsenic in these foods range from 0.02 mg/kg in grains and cereals to 0.14 mg/kg in meat, fish, and poultry (Gartrell et al. 1986). Shellfish and other marine foods contain the highest arsenic concentrations and are the largest dietary source of arsenic (Gunderson 1995a; Jelinek and Corneliussen 1977; Tao and Bolger 1999). Arsenic levels in various fish and shellfish are presented in Table 6-4. In the U.S. Food and Drug Administration (FDA) Total Diet Study, 1991–1997, seafood contained the highest levels of arsenic, followed by rice/rice cereal, mushrooms, and poultry. Concentrations in canned tuna (in oil), fish sticks, haddock (pan-cooked), and boiled shrimp were 0.609–1.470, 0.380–2.792, 0.510–10.430, and 0.290–2.681 mg/kg, respectively (Tao and Bolger 1999). Typically, arsenic levels in foods in the Total Diet Study, 1991–1996 were low, <0.03 mg/kg; only 63 of the 264 foods contained arsenic above this level. Similar results were reported in the Total Diet Study,

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Table 6-4. Levels of Arsenic in Fish and Shellfish—Recent Studies

Sample type	Arsenic concentration ^a (µg/g)	Comments	Reference
Yellowtail flounder		Samples collected from Northwest Atlantic 1993	Hellou et al. 1998
Muscle (n=8)	8–37		
Liver (n=6)	7–60		
Gonad (n=6)	1.2–9.4		
Marine organisms		Belgian fish markets in 1991; inorganic arsenic ranged from 0.003 to 0.2 µg/g	Buchet and Lison 1998
Ray (n=8)	16.4		
Cod (n=8)	4.7		
Plaice (n=8)	19.8		
Sole (n=8)	5.1		
Sea-bream (n=8)	2.4		
Mussell (n=8)	3.5		
Bluefin tuna (<i>Thunnus thynnus</i>) (n=14)	3.2	Virgin Rocks, Grand Banks of Newfoundland, Canada, 1990	Hellou et al. 1992
Fish		National Contaminant Biomonitoring Program, 1984–1985, 112 stations	Kidwell et al. 1995
Bottom feeding (n=2,020)	0.16±0.23 wet weight		
Predatory (n=12)	0.16±0.140 wet weight		
Oysters		South Carolina, private residential docks on tidal creeks, 1994	Wendt et al. 1996
<1 m from docks (n=10)	8.3±1.1		
>10 m from docks (n=10)	7.6±0.9		
Reference (no docks) (n=10)	8.4±1.3		
Clams (n=22)	12±1.1	Indian River Lagoon, Florida, 22 sites, 1990	Trocine and Trefry 1996
Marine organisms		Swan Lake, Galveston Bay, Texas, 1993	Park and Presley 1997
Snails	13.3±17.0		
Blue crab	6.61		
Fish	0.82		
Shrimp	1.37±0.64		
Whole crab	5.35±2.51		
Oysters (n=10, pooled)	7.28±1.32		
Mussels (n=7, pooled)	7.75±2.15		
Marine organisms		GPNEP, 1992, Galveston Bay, Texas	Park and Presley 1997
Blue crab	2.31±2.15		
Fish	2.46		
Oysters, two areas		NOAA NS&T Program, 1986–1990	Park and Presley 1997
n=78, pooled	4.50±1.08	Galveston Bay	
n=874, pooled	9.67±7.00	Gulf of Mexico	

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Table 6-4. Levels of Arsenic in Fish and Shellfish—Recent Studies

Sample type	Arsenic concentration ^a (µg/g)	Comments	Reference
Marine crustaceans			
<i>Parapenaeus longirostris</i> (pink shrimp) (n=826, 10 pools)	34.84±19.21 (12.01–62.60)	Commercial crustaceans from the Mediterranean Sea (Italy)	Storelli and Marcotrigiano 2001
<i>Aristeus antennatus</i> (red shrimp) (n=387, 8 pool)	17.09±3.49 (10.45–20.82)		
<i>Plesionika martia</i> (shrimp) (n=456, 7 pools)	40.82±2.50 (36.37–44.06)		
<i>Nephrops norvegicus</i> (Norway lobster) (n=270, 5 pools)	43.48±14.21 (35.63–69.15)		
Freshwater fish			
Sabalo (<i>Brycon melanopterus</i>) (n=3)	0.015–0.101	Fish samples (muscle) were collected in August 1997 from the Candamo River, Peru; a pristine rainforest valley prior to the start of oil-drilling activities	Gutleb et al. 2002
Huazaco (<i>Hoplias malabaricus</i>) (n=4)	nd–0.005		
Bagre (<i>Pimelodus ornatus</i>) (n=8)	nd–0.201		
Boquichio (<i>Prochilodus nigricans</i>) (n=1)	0.063		
Doncello (<i>Pseudo-platystoma sp.</i>) (n=1)	0.055		
Freshwater fish			
Bowfin (n=59)	0.32±0.04 wet weight	Savannah River, along and below the Department of Energy's Savannah River Site (SRS); samples analyzed were edible fillets	Burger et al. 2002
Bass (n=47)	0.03±0 wet weight		
Channel catfish (n=50)	0.09±0.02 wet weight		
Chain pickerel (n=19)	0.05±0.01 wet weight		
Yellow perch (n=51)	0.05±0.01 wet weight		
Black crappie (n=52)	0.04±0.01 wet weight		
American eel (n=24)	0.04±0.01 wet weight		
Shellcracker (n=52)	0.06±0 wet weight		
Bluegill (n=52)	0.05±0.02 wet weight		
Redbreast (n=43)	0.07±0.01 wet weight		
Spotted sucker (n=35)	0.03±0 wet weight		

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Table 6-4. Levels of Arsenic in Fish and Shellfish—Recent Studies

Sample type	Arsenic concentration ^a (µg/g)	Comments	Reference
Horseshoe crabs			
Apodeme (n=74)	7.034±0.65 wet weight	Overall mean in tissues of crabs collected from New Jersey in 2000	Burger et al. 2003
Egg (n=63)	5.924±0.345 wet weight		
Leg (n=74)	14.482±0.685 wet weight		
Apodeme (n=40)	7.513±0.835 wet weight	Overall mean in tissues of crabs collected from Delaware in 2000	
Egg (n=35)	6.766±0.478 wet weight		
Leg (n=40)	18.102±1.489 wet weight		

^aConcentrations are means±standard deviation, unless otherwise stated. Concentrations are in a dry weight basis, unless otherwise stated.

GM = geometric mean; GPNEP = Galveston Bay National Estuary Program; nd = not detected; NOAA NS&T = National Oceanic and Atmospheric Administration National Status and Trends

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1991–1997, where the mean arsenic concentration in all foods was 0.036 mg/kg dry weight and arsenic was not detectable in about 88% of the foods and was detected at trace levels in another 7.8% of foods. The foods with the highest mean arsenic levels were haddock, canned tuna, fish sticks, shrimp, and fish sandwiches, with arsenic concentrations ranging from 5.33 to 0.568 mg/kg dry weight (Capar and Cunningham 2000). Nriagu and Lin (1995) analyzed 26 brands of wild rice sold in the United States and found arsenic levels ranging from 0.006 to 0.142 $\mu\text{g/g}$ dry weight. Arsenic concentrations ranging from 0.05 to 0.4 $\mu\text{g/g}$ are typically reported for rice from North America, Europe, and Taiwan (Meharg and Rahman 2003).

During a comprehensive total diet study extending from 1985 to 1988, foods were collected in six Canadian cities and processed into 112 composite food samples (Dabeka et al. 1993). The mean, median, and range of total arsenic in all samples were 0.0732, 0.0051, and <0.0001–4.840 $\mu\text{g/g}$, respectively. Food groups containing the highest mean arsenic levels were fish (1.662 $\mu\text{g/g}$), meat and poultry (0.0243 $\mu\text{g/g}$), bakery goods and cereals (0.0245 $\mu\text{g/g}$), and fats and oils (0.0190 $\mu\text{g/g}$). Of the individual samples, marine fish had the highest arsenic levels, with a mean of 3.048 $\mu\text{g/g}$ for the cooked composites and 2.466 $\mu\text{g/g}$ for the raw samples. Canned fish (1.201 $\mu\text{g/g}$) and shellfish (2.041 $\mu\text{g/g}$) also contained high means. Cooked poultry, raw mushrooms, and chocolate bars contained 0.100, 0.084, and 0.105 $\mu\text{g/g}$, respectively.

National monitoring data from the Food Safety and Inspection Service National Residue Program (NRP) (1994–2000) found that the mean total arsenic concentration in livers of young chickens ranged from 0.33 to 0.43 $\mu\text{g/g}$, with an overall mean of 0.39 $\mu\text{g/g}$ (Lasky et al. 2004). The mean arsenic concentrations in liver for mature chickens, turkeys, hogs, and all other species over the same time period ranged from 0.10 to 0.16 $\mu\text{g/g}$. Lasky et al. (2004) used the NRP arsenic data in livers of young chickens to estimate the concentrations of arsenic in muscle tissue, the most commonly consumed part of the chicken. Assuming that 65% of the arsenic in poultry and meat is inorganic, at a mean level of chicken consumption of 60 g/person/day, people may ingest an estimated 1.38–5.24 $\mu\text{g/day}$ of inorganic arsenic from chicken.

A Danish study (Pedersen et al. 1994) reports the arsenic levels in beverages as the mean (range) in $\mu\text{g/L}$ as follows: red wine, 9 (<2–25); white wine, 11 (<2–33); fortified wine, 5 (<2–11); beer, 7 (4–11); soft drinks, 3 (<2–8); miscellaneous juices, 8 (3–13); instant coffee, 4 (0.7–7); and instant cocoa, 5.6 (1.6–12.8).

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In a study of dietary arsenic exposure in the Indigenous Peoples of the western Northwest Territories, Canada, fish contained the highest arsenic concentrations in foods consumed by the Dene and Métis populations with the highest concentration, 1.960 µg/g, found in smoked/dried cisco (fish). Other foods derived from land mammals, birds, and plants contained lower arsenic concentrations. A mean arsenic intake of <1.0 µg/kg/day was reported for this population (Berti et al. 1998).

The general consensus in the literature is that about 85–>90% of the arsenic in the edible parts of marine fish and shellfish is organic arsenic (e.g., arsenobetaine, arsenocholine, dimethylarsinic acid) and that approximately 10% is inorganic arsenic (EPA 2003b). However, the inorganic arsenic content in seafood may be highly variable. For example, a study in the Netherlands reported that inorganic arsenic comprised 0.1–41% of the total arsenic in seafood (Vaessen and van Ooik 1989). Buchet et al. (1994) found that, on the average, 3% of the total arsenic in mussels was inorganic in form. Some commercially available seaweeds, especially brown algae varieties, may have high percentages of the total arsenic present as inorganic arsenic (>50%) (Almela et al. 2002; Laparra et al. 2003). Arsenic concentrations ranging from 17 to 88 mg/kg dry weight were found in commercially available seaweeds (van Netten et al. 2000). Other arsenic compounds that may be found in seafood are arsenic-containing ribose derivatives called arsenosugars. Arsenosugars are the common organoarsenicals found in marine algae; they are also found in mussels, oysters, and clams (Le et al. 2004). Less information about the forms of arsenic in freshwater fish is known at this time (EPA 2003b).

Schoof et al. (1999a) reported on the analysis of 40 commodities anticipated to account for 90% of dietary inorganic arsenic intake. In this study, the amount of inorganic arsenic was measured in these foods. Consistent with earlier studies, total arsenic concentrations were highest in the seafood sampled (ranging from 160 ng/g in freshwater fish to 2,360 ng/g in marine fish). In contrast, average inorganic arsenic in seafood ranged from <1 to 2 ng/g. The highest inorganic arsenic concentrations were found in raw rice (74 ng/g), followed by flour (11 ng/g), grape juice (9 ng/g), and cooked spinach (6 ng/g).

Tobacco contains an average arsenic concentration of 1.5 ppm, or about 1.5 µg per cigarette (EPA 1998j). Before arsenical pesticides were banned, tobacco contained up to 52 mg As/kg, whereas after the ban, maximum arsenic levels were reduced to 3 µg/g (Kraus et al. 2000). An international literature survey reports arsenic yields of 0–1.4 µg/cigarette for mainstream (inhaled) cigarette smoke (Smith et al. 1997). The wide range of arsenic yields for flue-cured cigarettes suggests that the field history, soil, and fertilizer conditions under which the tobacco is grown will affect the arsenic concentration (Smith et al. 1997).

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Arsenic emission factors of 0.015–0.023 $\mu\text{g}/\text{cigarette}$ (mean $0.018 \pm 0.003 \mu\text{g}/\text{cigarette}$) have been measured for sidestream smoke from a burning cigarette (Landsberger and Wu 1995).

A median arsenic concentration of 2.1 $\mu\text{g}/\text{g}$ and a deposition rate of 0.008 $\mu\text{g}/\text{m}^2/\text{day}$ was reported in house dust in homes evaluated as part of the German Environmental Survey in 1990–1992. A mean arsenic concentration of 7.3 $\mu\text{g}/\text{g}$ was reported in house dust from 48 residences in Ottawa, Canada (Butte and Heinzow 2002). These arsenic concentrations are expected to be representative of background levels. In general high arsenic concentrations were found in household dust collected from homes in areas with known arsenic contamination. Mean arsenic concentrations of 12.6 (2.6–57) and 10.8 (1.0–49) $\mu\text{g}/\text{g}$ were reported in house dust collected from the entryway and child play areas, respectively, from homes in a community in Washington State with a history of lead arsenate use (Wolz et al. 2003). Arsenic was detected in all 135 indoor floor dust samples collected as part of the NHEXAS from Arizona mining communities, ranging from 0.3–50.6 $\mu\text{g}/\text{g}$, (O'Rourke et al. 1999). A geometric mean arsenic concentration of 10.8 $\mu\text{g}/\text{g}$ (range 1.0–172 $\mu\text{g}/\text{g}$) was reported in house dust from 96 homes in Middleport, New York, with historical pesticide manufacture, collected during the summer and fall of 2003 (Tsuji et al. 2005).

Arsenic has also been detected in several homeopathic medicines at concentrations up to 650 $\mu\text{g}/\text{g}$ (Kerr and Saryan 1986). Some Asian proprietary medicines that are manufactured in China, Hong Kong, and other Asian countries have been reported to contain levels of inorganic arsenic ranging from 25 to 107,000 $\mu\text{g}/\text{g}$ (Chan 1994). Fifty medicinally important leafy samples that were analyzed for elemental concentrations contained arsenic at levels ranging from 0.12 to 7.36 $\mu\text{g}/\text{g}$, with a mean of $2.38 \pm 1.2 \mu\text{g}/\text{g}$ (Reddy and Reddy 1997). Arsenic concentrations ranged from 0.005 to 3.77 $\mu\text{g}/\text{g}$ in 95 dietary supplements purchased from retail stores in the Washington, DC area in 1999 (Dolan et al. 2003). Commercially available samples of Valarian, St. John's Wort, Passion Flower, and Echinacea were purchased in the United States and analyzed for various contaminants; arsenic concentrations were 0.0016–0.0085, 0.0065–0.017.8, 0.0024–0.0124, and 0.0021–0.0102 $\mu\text{g}/\text{g}$, respectively, in these samples (Huggett et al. 2001). Concentrations of heavy metals including the metalloid arsenic were evaluated in 54 samples of Asian remedies that were purchased in stores in Vietnam and Hong Kong that would be easily accessible to travelers, as well as in health food and Asian groceries in Florida, New York, and New Jersey. Four remedies were found to contain daily doses exceeding 0.1 mg. Two of these contained what would have been a potentially significant arsenic dose, with daily doses of 16 and 7.4 mg of arsenic (Garvey et al. 2001).

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The possible presence of toxic compounds in waste materials has raised concerns about the fate of these compounds either during the composting process or when the composted product is applied to soils. Three waste compost products generated at the Connecticut Agricultural Experiment Station had arsenic levels of 12.8, 9.8, and 13 $\mu\text{g/g}$ dry weight, respectively (Eitzer et al. 1997). The arsenic levels in municipal solid waste composts from 10 facilities across the United States ranged from 0.9 to 15.6 $\mu\text{g/g}$ dry weight with a mean of 6.7 $\mu\text{g/g}$ (He et al. 1995). These are lower than the EPA 503 regulatory limit for arsenic of 41 $\mu\text{g/g}$ for agricultural use of sewage sludge (EPA 1993b). Concentrations of arsenic in U.S. sewage sludges, which are sometimes spread on soil, were <1 $\mu\text{g/g}$. Arsenic is a common impurity in minerals used in fertilizers. A comprehensive Italian study found that the arsenic content in a number of mineral and synthetic fertilizers ranged from 2.2 to 322 mg/kg with a sample of triple superphosphate having the highest level (Senesi et al. 1999). Arsenic naturally occurs in coal and crude oil at levels of 0.34–130 and 0.0024–1.63 ppm, respectively, which would account for its presence in flue gas, fly ash, and bottom ash from power plants (Pacyna 1987).

Background arsenic levels in living organisms are usually <1 $\mu\text{g/g}$ wet weight (Eisler 1994). Levels are higher in areas with mining and smelting activity or where arsenical pesticides were used. Eisler (1994) has an extensive listing of arsenic levels in terrestrial and aquatic flora and fauna from literature sources to about 1990. The U.S. Fish and Wildlife Service's National Contaminant Biomonitoring Program have analyzed contaminants in fish at 116 stations (rivers and the Great Lakes) across the United States. The geometric mean concentration of arsenic for the five collection periods starting in 1976 were (period, concentration wet weight basis): 1976–1977, 0.199 $\mu\text{g/g}$; 1978–1979, 0.129 $\mu\text{g/g}$; 1980–1981, 0.119 $\mu\text{g/g}$; 1984, 0.106 $\mu\text{g/g}$; and 1986, 0.083 $\mu\text{g/g}$ (Schmitt et al. 1999). In 1986, the maximum and 85th percentile arsenic levels were 1.53 and 0.24 $\mu\text{g/g}$, respectively. The highest concentrations of arsenic for all five collection periods were in bloaters from Lake Michigan at Sheboygan, Wisconsin. Arsenic levels declined by 50% at this site between 1976–1997 and 1984. The major source of arsenic into Lake Michigan was a facility at Marinette, Wisconsin, which manufactured arsenic herbicides. Table 6-4 contains arsenic levels in aquatic organisms from more recent studies. The Coeur d'Alene river basin in northern Idaho has been contaminated with heavy metals from mining and smelting operations since 1885 (Farag et al. 1998). A 1994 study determined the metal content of sediment, biofilm, and invertebrates at 13 sites in the basin, 10 with historic mining activity, and 3 reference sites. The mean arsenic levels in benthic macroinvertebrates at the mining sites ranged from 2.2 to 97.0 $\mu\text{g/g}$ dry weight, compared to 2.1–2.4 $\mu\text{g/g}$ dry weight at the reference sites. A study of aquatic organism in Swan Lake, a highly polluted sub-bay of Galveston Bay, Texas showed that arsenic concentrations were in the order snail>oyster>crab>shrimp>fish (Park and Presley 1997). In contrast to metals like silver, cadmium,

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copper, and zinc, arsenic concentrations in oysters and mussels were less than in the sediment from which they were collected. No significant correlation was found between levels of arsenic in clams in the Indian River Lagoon in Florida with those found in sediment or water samples (Trocine and Trefry 1996). Small animals living at mining sites ingest more arsenic in their diet and have higher arsenic levels in their bodies than those living on uncontaminated sites (Erry et al. 1999). Seasonal variations in both arsenic intake and dietary composition may affect the amount of arsenic taken up by the body and transferred to predator animals. Tissue arsenic content of wood mice and bank voles living on both arsenic-contaminated mining sites and uncontaminated sites were greater in autumn than spring. The lower tissue arsenic levels in spring of rodents living on contaminated sites suggest that there is no progressive accumulation of arsenic in overwintering animals.

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

Exposure to arsenic may include exposure to the more toxic inorganic forms of arsenic, organic forms of arsenic, or both. While many studies do not indicate the forms of arsenic to which people are exposed, this information may often be inferred from the source of exposure (e.g., fish generally contain arsenic as arsenobetaine). Yost et al. (1998) reported that the estimated daily dietary intake of inorganic arsenic for various age groups ranged from 8.3 to 14 $\mu\text{g}/\text{day}$ and from 4.8 to 12.7 $\mu\text{g}/\text{day}$ in the United States and Canada, respectively, with 21–40% of the total dietary arsenic occurring in inorganic forms.

Drinking water may also be a significant source of arsenic exposure in areas where arsenic is naturally present in groundwater. While estimates of arsenic intake for typical adults drinking 2 L of water per day average about 5 $\mu\text{g}/\text{day}$ (EPA 1982c), intake can be much higher (10–100 $\mu\text{g}/\text{day}$) in geographical areas with high levels of arsenic in soil or groundwater (see Figure 6-2). It is assumed that nearly all arsenic in drinking water is inorganic (EPA 2001).

In the United States, food intake of arsenic has been estimated to range from 2 $\mu\text{g}/\text{day}$ in infants to 92 $\mu\text{g}/\text{day}$ in 60–65-year-old men (see Table 6-5) (Tao and Bolger 1999). The average intake of inorganic arsenic are estimated to range from 1.34 $\mu\text{g}/\text{day}$ in infants to 12.54 $\mu\text{g}/\text{day}$ in 60–65-year-old men. Tao and Bolger (1999) assumed that 10% of the total arsenic in seafood was inorganic and that 100% of the arsenic in all other foods was inorganic. The greatest dietary contribution to total arsenic was seafood (76–96%) for all age groups, except infants. For infants, seafood and rice products contributed 42 and 31%, respectively. Adult dietary arsenic intakes reported for other countries range

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Table 6-5. Mean Daily Dietary Intake of Arsenic for Selected U.S. Population Groups

Mean daily intake ($\mu\text{g}/\text{kg}$ body weight/day)	Date of study		
	1984–1986 ^a	1986–1991 ^b	1991–1997 ^c
Provisional tolerable daily intake (PTDI) ^d	2.1	2.1	2.1
6–11 months	0.82	0.5	0.31
2 years	1.22	0.81	1.80
14–16 years, female	0.54	0.36	0.41
14–16 years, male	0.60	0.39	0.24
25–30 years, female	0.66	0.44	0.44
25–30 years, male	0.76	0.51	0.72
60–65 years, female	0.71	0.46	1.08
60–65 years, male	0.74	0.48	1.14

^aGunderson 1995a^bGunderson 1995b^cTao and Bolger 1999^dNo agreement has been reached on a maximum acceptable intake for total arsenic; the FAO/WHO has assigned a PTDI for inorganic arsenic of 2.1 $\mu\text{g}/\text{kg}$ body weight for adults. Data from FDA studies. FDA does not recommend daily intake levels for Arsenic.

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from 11.7 to 280 $\mu\text{g}/\text{day}$ (Tao and Bolger 1999). Schoof et al. (1999b) estimated that intake of inorganic arsenic in the U.S. diet ranges from 1 to 20 $\mu\text{g}/\text{day}$, with a mean of 3.2 $\mu\text{g}/\text{day}$. In contrast, these estimates of inorganic arsenic intakes are based on measured inorganic arsenic concentrations from a market basket survey.

The FDA conducted earlier Total Diet Studies in 1984–1986 and 1986–1991. For the sampling period of June 1984 to April 1986, the total daily intake of arsenic from foods was 58.1 μg for a 25–30-year-old male with seafood contributing 87% of the total (Gunderson 1995a). For the sampling period from July 1986 to April 1991, the total daily intake of arsenic from foods was lower, 38.6 μg for a 25–30-year-old male. Seafood again was the major source of arsenic, contributing 88% of the total (Gunderson 1995b). Results of the two Total Diet Studies for selected population groups are shown in Table 6-5. The Total Diet Study for the sampling period from September 1991 to December 1996, shows that arsenic, at ≥ 0.03 $\mu\text{g}/\text{g}$, was found in 55 (21%) of the 261–264 foods/mixed dishes analyzed. The highest concentrations again were found in seafood, followed by rice/rice cereal, mushrooms, and poultry. The estimated total daily intake of arsenic from foods was 56.6 μg for a 25–30-year-old male. Seafood was the major contributor, accounting for 88–96% of the estimated total arsenic intake of adults.

Average daily dietary exposures to arsenic were estimated for approximately 120,000 U.S. adults by combining data on annual diet, as measured by a food frequency questionnaire, with residue data for table-ready foods that were collected for the annual FDA Total Diet Study. Dietary exposures to arsenic were highly variable, with a mean of 50.6 $\mu\text{g}/\text{day}$ (range, 1.01–1,081 $\mu\text{g}/\text{day}$) for females and 58.5 $\mu\text{g}/\text{day}$ (range, 0.21–1,276 $\mu\text{g}/\text{day}$) for males (MacIntosh et al. 1997). Inorganic arsenic intake in 969 men and women was assessed by a semi-quantitative food frequency questionnaire in combination with a database for total arsenic content in foods and by toenail concentrations of arsenic. The mean estimated average daily consumption of inorganic arsenic was 10.22 $\mu\text{g}/\text{day}$ with a range of 0.93–104.89 $\mu\text{g}/\text{day}$. An assumption of 1.5% of the total arsenic in fish and 20% of the total arsenic in shellfish was inorganic arsenic was used in this assessment (MacIntosh et al. 1997).

During a comprehensive total diet study extending from 1985 to 1988, the estimated daily dietary ingestion of total arsenic by the average Canadian was 38.1 μg and varied from 14.9 μg for the 1–4 year-old-age group to 59.2 μg for 20–39-year-old males (Dabeka et al. 1993). Daily intakes of arsenic from food by women in the Shiga Prefecture, Japan, were investigated by the duplicate portion method and by the market basket method. In 1991 and 1992, the daily intakes determined by the duplicate portion

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method were 206 and 210 μg , respectively. Those determined by the market basket method were 160 and 280 μg , respectively (Tsuda et al. 1995b).

Arsenic concentrations in human breast milk have been reported to range from 4 to <10 $\mu\text{g/L}$ in pooled human milk samples from Scotland and Finland to 200 $\mu\text{g/L}$ in samples from Antofagasta, Chile, where there is a high natural environmental concentration of arsenic (Broomhall and Kovar 1986). The arsenic concentration in the breast milk of 35 women in Ismir, Turkey, a volcanic area with high thermal activity ranged from 3.24 to 5.41 $\mu\text{g/L}$, with a median of 4.22 $\mu\text{g/L}$ (Ulman et al. 1998). Sternowsky et al. (2002) analyzed breast milk from 36 women from three different regions in Germany. These regions included the city of Hamburg, a rural area, Soltau, Lower Saxony, and Munster, the potentially contaminated area. Arsenic was not detected (<0.3 $\mu\text{g/L}$) in 154 of 187 samples, with the highest concentration, 2.8 $\mu\text{g/L}$, found in a sample from the rural area. The geometric means from the three areas were comparable.

The mean arsenic levels in three groups of cows in the region that grazed on land impacted by lava and thermal activity were 4.71, 4.46, and 4.93 $\mu\text{g/L}$, compared to 5.25 $\mu\text{g/L}$ for cows kept in sheds and fed commercial pellet feed and municipal water (Ulman et al. 1998). Mean arsenic concentrations in cow's milk ranging from 18.6 to 17.1 $\mu\text{g/L}$ and from 16.7 to 18.0 $\mu\text{g/L}$ were reported for cow's grazing in nonindustrial and an industrial regions, respectively, in Turkey (Erdogan et al. 2004).

A Danish study found that carrots grown in soil containing 30 $\mu\text{g/g}$ of arsenic, which is somewhat above the 20 $\mu\text{g/g}$ limit for total arsenic set by Denmark for growing produce, contained 0.014 $\mu\text{g/g}$ fresh weight of arsenic, all in the form of inorganic As(III) and As(V) (Helgesen and Larsen 1998). An adult consuming 376 grams of vegetables a day (90th percentile) represented solely by carrots would consume 5.3 μg of arsenic a day. The study concluded that the estimated intake of arsenic from produce grown in soil meeting regulatory limits was low compared with other food sources and water.

If vegetables are grown in planters made of wood treated with CCA, arsenic may leach out of the wood and be taken up by the vegetables. In a study by Rahman et al. (2004), arsenic was found to diffuse into the soil from the CCA-treated wood, with the highest concentrations found at 0–2 cm from the CCA-treated wood and a steady decline in concentration with increased distance from the wood. Crops grown within 0–2 cm of the CCA-treated wood contained higher concentrations of arsenic than those grown at 1.5 m from the treated wood. However, the concentrations are below U.S. FDA tolerance limits that have been set for arsenic in select food items. In addition, food grown in this manner is unlikely to constitute a significant part of a person's diet.

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In 2003, U.S. manufacturers of arsenical wood preservatives began a voluntary transition from CCA to other wood preservatives in wood products for certain residential uses, such as play structures, picnic tables, decks, fencing, and boardwalks. This phase out was completed on December 31, 2003; wood treated prior to this date could still be used and structures made with CCA-treated wood would not be affected. CCA-treated wood products continue to be used in industrial applications (EPA 2003a). EPA's Consumer Awareness Program (CAP) for CCA is a voluntary program established by the manufacturers of CCA products to inform consumers about the proper handling, use, and disposal of CCA-treated wood. Additional information about this program can be found from EPA (2007a).

The arsenic content in the human body is 3–4 mg and tends to increase with age. Arsenic concentrations in most tissues of the human body are <0.3 to 147 µg/g dry weight, excluding hair, nails, and teeth. Mammals tend to accumulate arsenic in keratin-rich tissues such as hair and nails. The normal concentrations of arsenic range from about 0.08 to 0.25 µg/g in hair, and 0.34 µg/g in nails. The normal concentration of arsenic in urine can range from 5 to 40 µg per day (total) (Mandal and Suzuki 2002). Table 6-6 contains arsenic levels in various human tissues.

A German study investigated the transfer of arsenic from the environment to humans in the northern Palatine region, a former mining area characterized by high soil levels of arsenic (<2–605 µg/g) in residential areas compared to a region in southern lower Saxony with nonelevated levels of arsenic in soil (Gebel et al. 1998a). None of the residents were occupationally exposed to arsenic and the arsenic levels in drinking water were generally below 0.015 mg/L. The mean levels of arsenic in urine and hair were lower in the reference area than in the former mining area (see Table 6-6), although within the mining area, there was a slight increase in arsenic levels in hair and arsenic excreted in urine with increasing arsenic content in soil. Children in the Palatine region did not have higher contents of arsenic in their hair or urine. The most significant factor contributing to elevated levels of arsenic in hair and urine was seafood consumption. In the combined population of people living in mining areas containing high levels of arsenic in soil and other areas, the level of arsenic in urine was positively associated with the extent of seafood consumption. However, the study also showed that seafood consumption does not lead to an extreme increase in excretion of arsenic in the urine. There are apparently other, unidentified factors affecting the urine levels. Only arsenic in urine, not in hair, was significantly correlated with age. The level of arsenic in urine was very slightly, but significantly correlated with the consumption of home-grown produce. Tobacco smoking had no correlation with the arsenic content of either hair or urine (Gebel et al. 1998a).

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Table 6-6. Levels of Arsenic in Human Tissue and Urine—Recent Studies

Site population	Sample	Concentration		Units	Reference
		Mean ^a	Range		
Fort Valley, Georgia, Pesticide manufacturing facility (Superfund site)					
40 workers (samples collected at end of work week)	Urine, random	11.6	<1–57	µg/L	Hewitt et al. 1995
	Urine, 24-hour	11.0	<1–54	µg/L	
	Hair	0.78	<0.01–6.3	µg/g	
	Fingernails	0.79	<0.01–6.1	µg/g	
Hermosa, Sonora, Mexico					
Children, ages 7–11, exposed to arsenic in water (mean concentration [mean dose]):	Urine, 24-hour				Wyatt et al. 1998a, 1998b
	9 µg/L [0.481 µg/kg/day]	10.26	4.05–19.68	µg/day	
	15 µg/L [0.867 µg/kg/day]	10.54	2.82–20.44	µg/day	
	30 µg/L [1.92 µg/kg/day]	25.18	5.44–93.28	µg/day	
Glasgow, Scotland					
Adults, normal (n=1,250)	Hair	0.650	0.20–8.17	µg/g	Raie 1996
Adults, postmortem (n=9)	Liver	0.048 [0.024]	0.011–0.152	µg/g	
Infants, postmortem (n=9)	Liver	0.0099 [0.007]	0.0034–0.019	µg/g	
Adults, postmortem (n=8)	Lung	0.044 [0.022]	0.0121–0.125	µg/g	
Infants, postmortem (n=9)	Lung	0.007 [0.0055]	0.0011–0.015	µg/g	
Adults, postmortem (n=9)	Spleen	0.015 [0.008]	0.001–0.063	µg/g	
Infants, postmortem (n=8)	Spleen	0.0049 [0.0045]	0.0011–0.0088	µg/g	
Palatinate Region, Germany (high As) ^b					
Residents (n=199)	Urine, 24-hour	3.96 [3.21]	<0.1–18.32	µg/g	Gebel et al. 1998a
Residents (n=211)	Hair	0.028 [0.016]	<0.005–0.154	µg/g	
Saxony, Germany (low As—reference) ^b					
Residents (n=75)	Urine, 24-hour	7.58 [6.20]	0.29–23.78	µg/g	Gebel et al. 1998a
Residents (n=74)	Hair	0.069 [0.053]	0.013–0.682	µg/g	
Ismir, Turkey, (volcanic area with high thermal activity)					
Nonoccupationally exposed women (n=35)	Breast milk	4.23 [4.26]	3.24–5.41	µg/L	Ulman et al. 1998
Erlangen-Nuremberg Germany 1/92–12/93					
Nonoccupationally exposed people (n=50)	Lung	5.5	<1–13.0	ng/g ww	Kraus et al. 2000
		28.4	<1–73.6	ng/g dw	

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Table 6-6. Levels of Arsenic in Human Tissue and Urine—Recent Studies

Site population	Sample	Concentration		Units	Reference
		Mean ^a	Range		
Tarragona (Catalonia, Spain) 1997–1999					
Nonoccupationally exposed people (n=78)	Lung	<0.05		µg/g ww	Garcia et al. 2001
	Bone	<0.05			
	Kidney	<0.05			
	Liver	<0.05			
	Lung	<0.05			
West Bengal, India					
Residents consuming arsenic-contaminated water (n=47)	Fingernail	7.32	2.14–40.25	µg/g	Mandal et al. 2003
	Hair	4.46	0.70–16.17		
Residents consuming nonarsenic-contaminated water (n=15)	Fingernail	0.19	0.11–0.30		
	Hair	0.07	0.03–0.12		
Middleport, NY, USA					
Children <7 years (n=77)	Urine	15.1 ^c	2.1–59.6	µg/L	Tsuji et al. 2005
Children <13 years (n=142)	Urine	15.7 ^c	2.1–59.9		
Children ≥7 years and adults (n=362)	Urine	15.8 ^c	3.9–773		
All participants	Urine	15.7 ^c	2.1–773		

^aMedians, if reported, are in brackets.

^bThe reference group (Saxony) had significantly higher levels of arsenic in urine and hair. However, data from both groups correspond to normal range reference data.

^cGeometric mean, total arsenic

dw = dry weight; ww = wet weight

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A study was performed to look at the arsenic levels, as well as the arsenic species present, in hair and nail samples from individuals in an arsenic-affected area in West Bengal, India. Mean arsenic concentrations in hair and fingernails of the chronically arsenic exposed population were 4.46 and 7.32 $\mu\text{g/g}$, respectively and were 0.07 and 0.19 $\mu\text{g/g}$ in a control population. Fingernail samples were found to contain mostly inorganic arsenic (>80%) as a mixture of As(III) and As(V), as well as DMA(III) and DMA(V). Hair samples also mostly contained inorganic arsenic (>90%), as well as MMA(V) and DMA(V) (Mandal et al. 2003).

Arsenic in soil in communities surrounding former smelters is a public health concern, especially for infants and children who may consume significant quantities of soil. Since lead arsenate was used in apple and other fruit orchards, often at very high application rates, and this compound would be expected to accumulate and persist in surface soil, there are concerns to human health when these when old orchards are converted into subdivisions or when they are used to grow food crops or forage. However, arsenic in soil may be imbedded in minerals or occur as insoluble compounds such as sulfides and therefore, not be taken up by the body from the gastrointestinal tract. In addition, oxidation of mineral surfaces may result in armoring the primary mineral grain by a secondary reaction product. Arsenic-bearing solids are often encapsulated in insoluble matrices such as silica, further diminishing arsenic availability (Davis et al. 1992).

Sarkar and Datta (2004) examined the bioavailability of arsenic from two soils with different arsenic retention capacities. In this study, Immokalee (Florida) and Orelia (Texas) soils were incubated after spiking with sodium arsenate for 4 months. The Immokalee soil is a sandy spodosol with low Fe/Al, Ca/Mg, and P contents and is likely to have minimal arsenic retention capacity. The Orelia soil is a sandy clay that is expected to have strong arsenic retention capacity. Arsenic speciation and bioavailability were studied immediately after spiking and after 4 months of incubation. Approximately 85% of the total arsenic (soluble and exchangeable fractions) was considered bioavailable and phytoavailable immediately after pesticide application for the Immokalee soil; after 4 months of incubation, this decreased to approximately 46%. Immediately after pesticide application, the amounts of arsenic extracted in the soluble/exchangeable and Fe/Al-bound fractions were similar that of the Immokalee soil. After 4 months, the soluble arsenic decreased to approximately 45% and the Fe/Al-bound arsenic increased to about 40%. Experiments looking at the bioavailability of arsenic from these two soils indicated that the potentially irreversible adsorption of arsenic by the Orelia soil rendered a significant portion of the total arsenic unavailable for absorption by the human gastrointestinal system. Initially after pesticide application,

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100% of the arsenic was bioavailable; after 4 months, the bioavailable fraction was found to decrease to 88 and 69% in the Immokalee and Orelia soils, respectively (Sarkar and Datta 2004).

Hamel et al. (1998) used synthetic gastric juice to estimate the bioaccessible fraction of metals in the stomach with varying liquid to solid ratios. They found that the bioaccessibility may vary in different soils and with varying liquid to solid ratios. Bioaccessibility was defined as the amount of metal that is soluble in synthetic gastric juice and therefore, potentially available for uptake across the intestinal lumen, while bioavailability was defined as the amount that was actually taken across the cell membranes.

Arsenic bioaccessibility for National Institute of Standards and Technology (NIST) Montana Soil SRM 2710, with a certified arsenic concentration of 626 $\mu\text{g/g}$, was fairly consistent across the liquid-to-solid ratios and ranged from 41.8 ± 18 to $56 \pm 21\%$. The extractability of a hazardous waste contaminated soil from Jersey City, New Jersey, was different than that observed for the Montana NIST soil. For the Jersey City soil, which had an arsenic concentration of 1,120 $\mu\text{g/g}$, there was an increase in the bioaccessible arsenic as the liquid-to-solid ratio increased. Bioaccessible arsenic ranged from 4.5 ± 0.8 (at a liquid-to-solid ratio of 100:1) to $25 \pm 9\%$ (at a ratio of 5,000:1). Similarly, smelter impacted soils from Anaconda, Montana contain metal-arsenic oxides and phosphates whose bioaccessibility is limited by solubility restraints for residence times typical of the gastrointestinal tract (Davis et al. 1992, 1996).

Inhalation of arsenic from ambient air is usually a minor exposure route for the general population. For example, the dose to a person who breathes 20 m^3/day of air containing 20–30 ng/m^3 (see Section 6.4.1) would be about 0.4–0.6 $\mu\text{g}/\text{day}$. However, smokers may be exposed to arsenic by inhalation of mainstream smoke. Assuming that 20% of the arsenic in cigarettes is present in smoke, an individual smoking two packs of cigarettes per day would inhale about 12 μg of arsenic (EPA 1984a). However, a German study of the arsenic levels in lung tissue of 50 unexposed deceased people (see Table 6-6) found no significant difference in lung arsenic concentrations of smokers versus nonsmokers, nor were there any significant age- or sex-related differences (Kraus et al. 2000). Before arsenical pesticides were banned, tobacco contained up to 52 $\mu\text{g As/g}$, whereas after the ban, maximum arsenic levels were reduced to 3 $\mu\text{g/g}$.

Occupational exposure to arsenic may be significant in several industries, mainly nonferrous smelting, arsenic production, wood preservation, glass manufacturing, and arsenical pesticide production and application. Since arsenic compounds are used as a desiccant for cotton, workers involved in harvesting and ginning cotton may be exposed to arsenic. Occupational exposure would be via inhalation and dermal contact. Should any arsenic be retained in the cotton, workers handling the fabric and the general

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public would be exposed. The electronics industry is expanding the use of gallium arsenide in the production of electro-optical devices and integrated circuits, and workers in the industry where gallium arsenide is used may be exposed to hazardous substances such as arsenic, arsine, and various acids (Sheehy and Jones 1993). Occupational exposure to arsenic is generally assessed by measuring urinary excretion of arsenic. Past exposure is commonly assessed by arsenic levels in hair. Different types of occupational exposures may result in different uptakes of arsenic because of the bioavailability of the form of arsenic to which workers are exposed. For example, maintenance workers at a Slovak coal-fired power plant exposed to 8-hour TWA arsenic air concentrations of $48.3 \mu\text{g}/\text{m}^3$ (range, 0.17–375.2) had urinary total arsenic levels of $16.9 \mu\text{g As/g creatinine}$ (range, 2.6–50.8), suggesting that bioavailability of arsenic from airborne coal fly ash is about one-third that from in copper smelters and similar settings (Yager et al. 1997). Approximately 90% of the arsenic-containing particulates were $\geq 3.5 \mu\text{m}$. Apostoli et al. (1999) monitored 51 glass workers exposed to arsenic trioxide by measuring dust in the breathing zone. The mean concentration of arsenic in air was $82.9 \mu\text{g}/\text{m}^3$ (1.5–312 $\mu\text{g}/\text{m}^3$); exposure was higher for workers involved in handling the particulate matter. The occupational exposures to principal contaminants, including arsenic, at five coal-fired power plants were evaluated during June–August 2002. Eight air samples were collected per similar exposure group at four of the five facilities; inorganic arsenic concentrations in all samples were below the limit of detection (0.37–0.72 $\mu\text{g}/\text{m}^3$), as well as being below the OSHA permissible exposure limit (PEL) of 10 $\mu\text{g}/\text{m}^3$ (Bird et al. 2004).

NIOSH researchers conducted a study of arsenic exposures and control systems for gallium arsenide operations at three microelectronics facilities during 1986–1987 (Sheehy and Jones 1993). Results at one plant showed that in all processes evaluated but one, the average arsenic exposures were at or above the OSHA action level of 5 $\mu\text{g}/\text{m}^3$, with a maximum exposure of 8.2 $\mu\text{g}/\text{m}^3$. While cleaning the Liquid Encapsulated Czochralski (LEC) pullers, the average potential arsenic exposure of the cleaning operators was 100 times the OSHA PEL of 10 $\mu\text{g}/\text{m}^3$. Area arsenic samples collected at the plant in break-rooms and offices, 20–60 feet from the process rooms, had average arsenic concentrations of 1.4 $\mu\text{g}/\text{m}^3$. At the other two plants, personal exposures to arsenic were well controlled for all processes evaluated.

A study has been conducted to examine the relationship between total arsenic levels in hair of employees in a semiconductor fabrication facility and job responsibility, a surrogate variable for arsenic exposure (de Peyster and Silvers 1995). Airborne arsenic was found in areas where equipment was cleaned but not in administrative areas. The highest airborne arsenic level found in the study, 15 $\mu\text{g}/\text{m}^3$, was collected from the breathing zone of a maintenance employee who was cleaning a source housing over a period of 2 hours in an area with local exhaust ventilation. A concentration of 2 $\mu\text{g}/\text{m}^3$ was found during the

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remainder of the cleaning period (~53 minutes). Workers in maintenance who were regularly assigned to cleaning equipment, and therefore presumed to have the highest exposure potential, had a mean hair arsenic level of 0.042 $\mu\text{g/g}$. This was higher than the mean of 0.033 $\mu\text{g/g}$ observed in administrative controls, but the difference was not significant. Maintenance workers who only occasionally cleaned and maintained arsenic-contaminated equipment had a mean hair arsenic level of 0.034 $\mu\text{g/g}$, which was comparable to the controls. The highest group mean hair arsenic level of 0.044 $\mu\text{g/g}$, surprisingly, was found in supervisors and engineers who were presumed to have the lowest exposure potential of all workers in the process areas. However, the highest concentrations of hair arsenic in engineers, 0.076 and 0.106 $\mu\text{g/g}$, were observed in two heavy smokers who smoked 1–2 packs of cigarettes per day. A 2-way analysis of variance indicated that smoking appeared to be a significant contributing factor whereas occupational exposure was not.

Hwang and Chen (2000) evaluated arsenic exposure in 21 maintenance engineers (exposed group) and 10 computer programmers (control group) at 3 semiconductor manufacturing facilities. Samples of air, wipe, and urine, as well as used cleaning cloths and gloves were collected to determine arsenic exposure. Arsenic was undetectable in 46 of the 93 air samples, and most samples were generally below the recommended occupational exposure limit (10 $\mu\text{g}/\text{m}^3$) in work areas during ion implanter maintenance. Arsenic was detectable in 22 of the 45 area air samples and in 15 of the 35 personal air samples; however, all concentrations were well below the occupational exposure limit of 50 ppb (160 $\mu\text{g}/\text{m}^3$). Mean arsenic concentrations ranged from not detected to 4.0 ppb (15 $\mu\text{g}/\text{m}^3$) in area air samples, and the mean arsenic concentration of personal air for maintenance engineers was 4.3 ppb (14 $\mu\text{g}/\text{m}^3$). Arsenic concentrations in wipe samples, used cleaning cloths, and gloves, varied from not detected to 146 $\mu\text{g}/\text{cm}^2$. During ion implanter maintenance, urinary arsenic levels were found to increase (1.0–7.8 $\mu\text{g}/\text{g}$ creatinine) in the maintenance engineers, from a mean baseline concentration of 3.6 $\mu\text{g}/\text{g}$ creatinine. The average urinary arsenic level for the computer programmers was 3.8 $\mu\text{g}/\text{g}$ creatinine (Hwang and Chen 2000). Mean arsenic concentrations in blood of 103 workers in the optoelectronic industry and 67 controls were 8.58 and 7.85 $\mu\text{g}/\text{L}$, respectively (Liao et al. 2004).

Concentrations of various metals, including arsenic, were measured in autopsy tissues (liver, lung, kidney, brain, and bone) collected from 78 nonoccupationally exposed subjects from Tarragona County, Spain between 1997 and 1999. In general, arsenic concentrations were under the analytical detection limit (0.05 $\mu\text{g}/\text{g}$ wet weight) in all tissues (Garcia et al. 2001).

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CCA preservatives are commonly used for treating timber used in constructions in marine and other humid environments or in contact with the ground. Exposure to CCA compounds may occur through dermal contact and inhalation of dust while working with the treated timber. Nygren et al. (1992) investigated the occupational exposure to airborne dust, chromium, copper, and arsenic in six joinery shops in Sweden where impregnated wood was used for most of their production. The mean airborne concentration of arsenic around various types of joinery machines ranged from 0.54 to 3.1 $\mu\text{g}/\text{m}^3$. No increased concentrations of arsenic were found in the workers' urine. A study was carried out in Denmark to evaluate arsenic exposure in taxidermists, workers impregnating wood with CCA solutions, fence builders, construction workers, and workers impregnating electric pylons with arsenic solution (Jensen and Olsen 1995). Airborne arsenic exposure was documented in 19 of 27 individuals working with products containing arsenic. The maximum exposure concentration was 17.3 $\mu\text{g}/\text{m}^3$, found for a single worker who was filling an impregnation container with CCA paste. Median exposures for indoor workers producing garden fences and weekend cottages were 3.7 and 0.9 $\mu\text{g}/\text{m}^3$, respectively. The maximum urine concentration reported in the study was 294.5 nanomoles arsenic per millimole creatinine (195 $\mu\text{g As/g creatinine}$) and was from the injector impregnating electric pylons. The median concentration in workers on electric pylons was 80 nanomoles arsenic per millimole creatinine (53 $\mu\text{g As/g creatinine}$), which was 6 times the concentration in reference individuals. Urine arsenic levels in workers producing garden fences and in taxidermists were 2.9 and 1.8 times the reference level, respectively.

The NIOSH National Occupational Exposure Survey (NOES) conducted in 1981–1983 estimated that about 55,000 workers were potentially exposed to arsenic (NOES 1990). The NOES was based on field surveys of 4,490 facilities that included virtually all workplace environments, except mining and agriculture, where eight or more persons are employed. The principal exposure pathway is probably inhalation of arsenic adsorbed to particulates, but ingestion and possibly dermal exposure may also be common. Since arsenic is no longer produced in the United States (see Section 5.1) and many arsenical pesticide uses have been banned (see Chapter 8), it is likely that the number of workers occupationally exposed to arsenic has decreased markedly in more recent years.

6.6 EXPOSURES OF CHILDREN

This section focuses on exposures from conception to maturity at 18 years in humans. Differences from adults in susceptibility to hazardous substances are discussed in Section 3.7, Children's Susceptibility.

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Children are not small adults. A child's exposure may differ from an adult's exposure in many ways. Children drink more fluids, eat more food, breathe more air per kilogram of body weight, and have a larger skin surface in proportion to their body volume. A child's diet often differs from that of adults. The developing human's source of nutrition changes with age: from placental nourishment to breast milk or formula to the diet of older children who eat more of certain types of foods than adults. A child's behavior and lifestyle also influence exposure. Children crawl on the floor, put things in their mouths, sometimes eat inappropriate things (such as dirt or paint chips), and spend more time outdoors. Children also are closer to the ground, and they do not use the judgment of adults to avoid hazards (NRC 1993).

As with adults, most children are exposed to arsenic largely through their diet. Since the greatest dietary intake of arsenic is from fish and seafood, infants and young children for whom a substantial part of their food is milk, would not be exposed to arsenic from dietary sources as much as older children. Even when mothers consume large amounts of seafood, there does not appear to be any major transfer of arsenobetaine, the major form of arsenic in seafood, from seafood to milk (Grandjean et al. 1995). Arsenic concentrations were very low in human milk sampled from 88 mothers in the Faroe Islands, where the seafood diet includes pilot whale meat and blubber. The total arsenic concentrations ranged from 0.1 to 4.4 $\mu\text{g}/\text{kg}$, with a median of 1.6 $\mu\text{g}/\text{kg}$ (Grandjean et al. 1995). The arsenic concentration in the breast milk of 35 women in Ismir, Turkey, a volcanic area with high thermal activity ranged from 3.24 to 5.41 $\mu\text{g}/\text{L}$, with a median of 4.22 $\mu\text{g}/\text{L}$ (Ulman et al. 1998). The mean arsenic levels in three groups of cows in the region that grazed on land impacted by lava and thermal activity were 4.71, 4.46, and 4.93 $\mu\text{g}/\text{L}$, compared to 5.25 $\mu\text{g}/\text{L}$ for cows kept in sheds and fed commercial pellet feed and municipal water. The arsenic levels in the urine of pregnant women and the cord blood of their infants were 0.625 ± 0.027 and 0.825 ± 0.079 $\mu\text{g}/\text{L}$, respectively. The authors concluded that there was no harmful exposure to arsenic in volcanic areas with high arsenic levels from suckling infants or feeding them local cow's milk, nor was there harm to the newborns from their mother's diet. Sternowsky et al. (2002) analyzed breast milk from 36 women from three different regions in Germany. These regions included the city of Hamburg, a rural area, Soltau, Lower Saxony, and Munster, the potentially contaminated area. Arsenic was not detected (<0.3 $\mu\text{g}/\text{L}$) in 154 of 187 samples, with the highest concentration, 2.8 $\mu\text{g}/\text{L}$, found in a sample from the rural area. The geometric mean arsenic concentrations from the three areas were comparable. Calculated oral intakes of arsenic were between 0.12 and 0.37 $\mu\text{g}/\text{day}$ for an infant at 3 months of age and weighing 6 kg.

According to the FDA study of 1986–1991, the mean daily intakes of arsenic are 0.5 and 0.81 $\mu\text{g}/\text{kg}$ body weight per day for a 6–11-month-old infant and 2-year-old child, respectively (Gunderson 1995b). This

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can be compared to a mean daily intake of 0.51 $\mu\text{g}/\text{kg}$ -body weight per day for a 25–30-year-old male (see Table 6-5). A Total Diet Study, from September 1991 to December 1996, estimated that the average inorganic arsenic intake for children of various age/sex groups were (age-sex group, total arsenic intake in $\mu\text{g}/\text{day}$, inorganic arsenic intake in $\mu\text{g}/\text{day}$): 6–11 months, 2.15, 1.35; 2 years, 23.4, 4.41; 6 years, 30.3, 4.64; 10 years, 13.3, 4.21; and 14–16 years (females), 21.8, 5.15; 14–16 years (males), 15.4, 4.51 (Tao and Bolger 1999). The greatest dietary contribution (76–96%) of total arsenic intake for all age groups other than infants was seafood. For infants, 41 and 34% of the estimated total arsenic intakes are from seafood and rice/rice cereals, respectively (Tao and Bolger 1999). Only for toddlers does the intake approach the World Health Organization's (WHO) provisional tolerable daily intake (PTDI) for inorganic arsenic (see Table 6-5). A 1985–1988 Canadian total diet study estimated that 1–4-year-olds ingested 14.9 μg of total arsenic per day compared with 38.1 μg by the average Canadian and 59.2 μg for 20–39-year-old males (Dabeka et al. 1993). Yost et al. (2004) estimated the mean dietary intake for inorganic arsenic for children (1–6 years of age) to be 3.2 $\mu\text{g}/\text{day}$, with a range of 1.6–6.2 $\mu\text{g}/\text{day}$ for the 10th and 95th percentiles, respectively. Inorganic arsenic intake was predominantly contributed by grain and grain products, fruits and fruit juices, rice and rice products, and milk (Yost et al. 2004). Total arsenic and arsenobetaine concentrations were measured in 16 baby food samples obtained from manufactures in Spain; total arsenic concentrations ranged from 2.042 to 0.270 $\mu\text{g}/\text{g}$ in plaice with vegetables and sole with white sauce, respectively. Arsenobetaine, which is the arsenical commonly found in fish, accounted for essentially 100% of the arsenic present in the samples (Vinas et al. 2003).

Arsenic exposure from drinking water may be elevated especially in groundwater from areas where arsenic occurs naturally in soil such as the western and north central sections of the United States (see Table 6-3 and Figure 6-2).

Arsenic exposure in communities near mining and smelting facilities or where arsenic had formerly been applied to agricultural land are a public health concern, especially for infants and children. Since arsenic remains in the surface soil indefinitely and long past land uses may be forgotten, people may not realize that they are living in areas where high levels of arsenic may occur in soil. Contaminated soils pose a particular hazard to children because of both hand-to-mouth behavior and intentional ingestion of soil (pica) that contains metals and other contaminants (Hamel et al. 1998). In these communities, arsenic may contaminate carpeting or may have been tracked in from outside. Children may be exposed to this arsenic while crawling around or playing on contaminated carpeting. Exposure may also result from dermal contact with the soil, or by inhaling the dust and then swallowing it after mucociliary transport up

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out of the lungs. Because much of the arsenic in soil is embedded in or adsorbed to soil particles or insoluble, it may not be in a form accessible for uptake by the body.

Hwang et al. (1997b) studied the arsenic exposure of children in Anaconda, Montana, in the vicinity of a former copper smelter from the summer of 1992 through the summer of 1993. Environmental samples and first morning voided urine samples from 414 children <72 months old were collected. Attention was focused on that fraction of the environmental source that was thought to be of the greatest risk to the child (i.e., arsenic in small particles [$<250 \mu\text{m}$]) that could most readily adhere to hands and toys and could be inadvertently ingested. Average arsenic levels in different types of soil ranged from 121 to 236 $\mu\text{g/g}$. Several studies have reported mean soil ingestion values for children ranging from 9 to 1,834 $\mu\text{g/day}$. Assuming that high arsenic exposure areas have average arsenic levels in soil from 60 to 150 $\mu\text{g/g}$, the resulting daily arsenic intake from soil could range from 1 to 275 $\mu\text{g/day}$ per child. The geometric mean of speciated urinary arsenic (combined As(III), As(V), MMA and DMA) was $8.6 \pm 1.7 \mu\text{g/L}$ ($n=289$) in the Hwang study. A nationwide survey on arsenic exposure in the vicinity of smelter sites revealed that children without excess arsenic exposure had average total urinary arsenic levels ranging from 5 to 10 $\mu\text{g/L}$ (Hwang et al. 1997a). Compared to these values, the mean total urinary arsenic values found in the Hwang study were markedly higher, but they were still well below the WHO-recommended maximum excretion level for total arsenic of 100 $\mu\text{g/L}$ as an action level for intervention. The investigators hypothesized that the relatively low urinary arsenic levels found in the study were probably a reflection of the low bioavailability of some forms of arsenic in contaminated soil. Hwang et al. (1997a) stated that arsenic intake through skin contact is insignificant and may be neglected in the assessment of childhood arsenic exposure. They recommend that parents or guardians pay more attention to their children's activity, especially hand-to-mouth behavior, even though the environmental contaminants might be elevated only slightly. Children in the northern Palatine region of German study, a former mining area characterized by high levels of arsenic ($<2\text{--}605 \mu\text{g/g}$) in residential areas did not show higher arsenic levels in their hair or urine than children from a reference area of Germany (Gebel et al. 1998a).

While CCA registrants voluntarily canceled the production of CCA-treated wood for residential use in 2003, there is a potential for exposure to arsenic from existing structures (Zartarian et al. 2006). Based on a review of existing studies, Hemond and Solo-Gabriele (2004) estimated that children with contact with CCA-treated wood may be subjected to doses in the range of tens of micrograms of arsenic per day. The most important route of exposure appeared to be by hand-to-mouth activities after contact with the CCA-treated wood. Kwon et al. (2004) compared the amounts of water-soluble arsenic on hands of children in

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contact with CCA-treated wood structures or sand in playgrounds. The mean amount of water-soluble arsenic on children's hands from playgrounds without CCA-treated wood was 0.095 μg (range 0.011–0.41 μg). A mean amount of water-soluble arsenic on children's hands from playgrounds with CCA-treated wood was 0.5 μg (range 0.0078–3.5 μg) (Kwon et al. 2004). Additional data from the study by Kwon et al. (2004) showed that total arsenic collected in hand-washing water (insoluble arsenic on the filter combined with the water-soluble arsenic in the filtrate) was 0.934 and 0.265 μg for the CCA playgrounds and the non-CCA playgrounds, respectively (Wang et al. 2005). Two wood surface swab samples collected from 217 play structures constructed from CCA-treated wood in the City of Toronto, Canada were sampled and analyzed for inorganic arsenic (Ursitti et al. 2004). Dislodgeable arsenic concentrations were found to vary widely from nondetectable (0.08–0.25 $\mu\text{g}/100\text{ cm}^2$) to 521 $\mu\text{g}/100\text{ cm}^2$ (mean = 41.6 $\mu\text{g}/100\text{ cm}^2$), and were found to not be a useful predictor of soil arsenic levels (Ursitti et al. 2004).

Shalat et al. (2006) evaluated postexposure hand rinses and urine for total arsenic for 11 children (13–71 months) in homes in Miami-Dade County, Florida, with and without CCA-treated playgrounds. Seven playgrounds were included in this study, and five of these contained either CCA-treated or partially CCA-treated wood. In addition, samples of wood, soil (5–8 cm from the base of the playground structure), mulch (when present), and synthetic wipes were analyzed for total arsenic. Wood and soil arsenic concentration were <2.0 and <3 mg/kg for the non-CCA-treated playgrounds, respectively. Mean arsenic concentrations of 2,380 mg/kg (range 1,440–3,270 mg/kg) and 19 mg/kg (4.0–42 mg/kg) were reported for wood and soil, respectively, in the playgrounds with CCA-treated wood. An arsenic concentration in mulch at one playground without CCA-treated wood was 0.4 mg/kg, and arsenic concentrations were 0.6 and 69 mg/kg in mulch at two of the playgrounds with CCA-treated wood. The amount of arsenic removed by synthetic wipes from the non-CCA-treated wood was <0.5 μg , while the mean amount of arsenic removed from the CCA-treated wood was 117 μg (range 1.0–313 μg). The amount of arsenic in hand rinses from children who played at the playgrounds with non-CCA-treated wood and at the playgrounds with CCA-treated wood were <0.2 and 0.6 μg (range <0.2–1.9 μg), respectively. The mean urinary total arsenic concentration was 0.0136 $\mu\text{g}/\text{L}$ (range 0.0072–0.0231 $\mu\text{g}/\text{L}$) for all children. No association between assess to CCA-treated playgrounds and urinary arsenic levels was found (Shalat et al. 2006).

The potential exposure children may receive from playing in play structures constructed from CCA-treated wood is generally smaller than that they would receive from food and water. For comparison, Yost et al. (2004) estimated the mean dietary intake for inorganic arsenic for children (1–6 years of age)

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to be 3.2 µg/day, with a range of 1.6–6.2 µg/day for the 10th and 95th percentiles, respectively. In a Total Diet Study, from September 1991 to December 1996, estimated average total intakes for children aged 6–11 months, 2 years, 6 years, and 10 years were 2.15, 23.4, 30.3, and 13.3 µg/day, respectively. Average inorganic arsenic intakes for the same age groups were estimated as 1.35, 4.41, 4.64, and 4.21 µg/day, respectively, based on data for total arsenic in foods and the assumption that 10% of the total arsenic in seafood was inorganic and that 100% of the arsenic in all other foods was inorganic (Tao and Bolger 1999). Hand washing after play would reduce the potential exposure to children to arsenic after playing on play structures constructed with CCA-treated wood, since most of the arsenic on the children's hands was removed with water (Kwon et al. 2004).

Concentrations of several toxic metals, including the metalloid arsenic, were measured in the placentas of 200 women in two urban cities in Ukraine, Kyiv and Dniprodzerzhinsk. Arsenic was detected in only 5% of the samples with concentrations ranging from <0.156 to 0.378 µg/g. In a study in Bulgaria, placental arsenic concentrations of 7 and 23 µg/g were reported in a control and smelter area, respectively. A placental arsenic concentration of 34 µg/g was reported in a region of Argentina with high concentrations of arsenic in drinking water (Zadorozhnaja et al. 2000).

Parents can inadvertently carry hazardous materials home from work on their clothes, skin, hair, and tools, and in their vehicles (DHHS 1995). Falk et al. (1981b) reported a case of hepatic angiosarcoma in a child that could be associated with arsenic contamination of a parent's clothing, the water supply, and the environment. The father worked in a copper mine and smelter area where his clothing was contaminated with dust containing arsenic. His daughter, who exhibited a high degree of pica, ate soil from the yard, and licked soil off her father's shoes. In a study of arsenic levels in homes in Hawaii, Klemmer et al. (1975) found higher levels in homes of employees of firms that used arsenic for pesticides or wood preservation, compared to homes where residents' work did not involve arsenic. The concentration of arsenic in dust from the homes of workers exposed to arsenic ranged from 5.2 to 1,080 µg/g, compared to concentrations of 1.1–31 µg/g in dust from control homes.

While the harmful effects of many components of tobacco smoke are well known, those due to heavy metals in the smoke have not been sufficiently emphasized. The adverse health effects of these toxic metals on the fetus through maternal smoking are of special concern (Chiba and Masironi 1992). The concentration of arsenic in tobacco is relatively low, usually below detectable limits (<1 µg/g). Although the concentrations of inorganic and organic arsenic in the urine of adults do not appear to be influenced by smoking, a positive association was found between urinary arsenic levels in children and parental

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smoking habits. As detailed in a WHO report, the mean arsenic level in the urine of children of nonsmoking parents was 4.2 $\mu\text{g/g}$ creatinine, in children with one smoking parent, it was 5.5 $\mu\text{g/g}$, and in children with both parents smoking, it was 13 $\mu\text{g/g}$ (Chiba and Masironi 1992). Tsuji et al. (2005) reported geometric mean concentrations of total arsenic of 15.1 and 15.7 $\mu\text{g/L}$ in children <7 and <13 years old, respectively, from households in Middleport, New York, where historical pesticide manufacture was associated with arsenic in soil. Geometric mean concentrations of inorganic arsenic, MMA, and DMA were 0.81, 0.54, and 2.5 $\mu\text{g/L}$, respectively, in children <7 years old and 0.83, 0.55, and 3.0 $\mu\text{g/L}$, respectively, in children <13 years old (Tsuji et al. 2005).

The use of Chinese herbal medicines (CHM) appears to be common among Chinese women. Both CHM and Chinese proprietary medicines (CPM) are used for treatment of minor ailments in babies and children. Herbal medicines are available in capsule or tablet form in drug stores, supermarkets, and by mail. The CPM “Sin Lak Pill,” “Lu Shen Wan,” and other anti-asthma preparations have been found to contain inorganic arsenic levels ranging from 25 to 107,000 $\mu\text{g/g}$, and cases of acute arsenic poisoning have been found in children and adults using these CPM (Chan 1994). Babies and children are particularly at risk because they may be given higher doses of these preparations per kg of body weight than adults would normally consume. They may also lack the hepatic enzymes responsible for drug biotransformation and detoxification (Chan 1994). Concentrations of heavy metals, including arsenic, were evaluated in 54 samples of Asian remedies that were purchased in stores in Vietnam and Hong Kong that would be easily accessible to travelers, as well as in health food and Asian groceries in Florida, New York, and New Jersey. One remedy that was recommended to treat children’s fever would expose a 15 kg child to approximately 5.0 mg of arsenic per day (Garvey et al. 2001). A folk remedy, purchased in California, for the treatment of chicken pox, flu-like symptoms, and nasal congestion, which had been given to two children in Wisconsin, was found to contain 36% arsenic acid. One-half teaspoon of this powder (about 500 mg of arsenic) was dissolved in hot water and taken 2–3 times per day (Werner et al. 2001).

Various metallic pigments and colors in the form of salts or lakes are used in toy production. Therefore, children may be exposed to toxic metals while playing with toys, especially when they lick, suck, or swallow a toy or a piece of a toy. Toys produced in European Union Markets must conform to restrictions concerning the bioavailability of toxic metals, including arsenic. The maximum limit for bioavailability of arsenic from the accessible parts of a toy is set to 0.1 $\mu\text{g/day}$. This corresponds to an arsenic migration limit of 25 $\mu\text{g/g}$ for all toy material, including modeling clay and paints (Rastogi and Pritzl 1996). A study was carried out to determine whether crayons, water colors, and water-based paints

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conform with the migration limits for toxic metals (Rastogi and Pritzl 1996). For the analysis, 94 samples representing 48 products were obtained from China, Taiwan, Japan, the United States, and European countries. Fifty-two samples showed migration of arsenic, ranging from 0.01 to 3.75 $\mu\text{g/g}$.

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

In addition to individuals who are occupationally exposed to arsenic (see Section 6.5), there are several groups within the general population that have potentially high exposures (higher than background levels) to arsenic. These populations include individuals living in proximity to sites where arsenic was produced, used (e.g., as a pesticide), or disposed, and individuals living near one of the 1,684 NPL hazardous waste sites where arsenic has been found at elevated levels in some environmental media (HazDat 2006). It also includes point sources such as smelters, coal-fired power plants, and municipal incinerators. People living in areas of volcanic activity may be exposed to higher levels of arsenic since high levels are more likely to be present in the environment. Other populations at risk of potentially high levels of exposure include those whose water supply contains high levels of arsenic and those consuming large amounts of seafood or seaweed. However, as pointed out previously (see Section 6.4.4), arsenic in fish and shellfish, is largely in the form of the less harmful organic arsenical, arsenobetaine; however, some commercially available seaweeds, especially brown algae varieties, may have high percentages of the total arsenic present as inorganic arsenic (>50%) (Almela et al. 2002; Laparra et al. 2003). While elevated urinary arsenic excretion levels have been associated with the consumption of fish and seafood, in a study of 32 sport fish consumers from Lakes Erie, Huron, and Michigan, only 6 (19%) had detectable urine arsenic concentrations, >4 $\mu\text{g/L}$, and 5 of these consumed fish from Lake Huron (Anderson et al. 1998). Exposure of high levels of arsenic in drinking water is more apt to be absorbed by the body and be harmful than exposure to arsenic in seafood. For example, a group of 36 people in Zimapán, Mexico who consumed water from an aquifer with 1.0 mg As/L had hair arsenic levels of 2.6–14.1 $\mu\text{g/g}$ (10 $\mu\text{g/g}$ average), compared with 2.4–13.9 $\mu\text{g/g}$ (6.19 $\mu\text{g/g}$ average) for a reference population that consumed bottled water with <0.014 mg/L arsenic (Armienta et al. 1997).

A study was conducted to determine if significant arsenic exposure was occurring at a Superfund site in Fort Valley, Georgia (Hewitt et al. 1995). Random urine, 24-hour urine, hair, and fingernail samples were collected at the end of the workweek from 40 employees at an active pesticide manufacturing facility where arsenical pesticides had been produced for over 50 years prior to the mid-1970s. Measurement of arsenic in the urine is considered to be the best method for monitoring recent exposure in industrial populations. Hair and fingernail analyses may provide an indication of exposures that occurred

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up to several months prior to testing, but both can adsorb and strongly retain arsenic from external sources. Since arsenic is rapidly cleared from the blood (half-life of 3–4 hours), blood arsenic levels are not considered suitable for monitoring populations for chronic low-level arsenic exposure. Results of the Hewitt study are summarized in Table 6-6. Urinary arsenic levels for all workers were well within the commonly accepted normal range of <100 µg/L.

As noted above, workers in a number of industries may have high exposures to arsenic, especially if proper safety procedures are not followed. For members of the general population, above-average exposure to arsenic from drinking water is possible in areas of high natural arsenic levels in groundwater or elevated arsenic levels in drinking water due to industrial discharges, pesticide applications, or leaching from hazardous waste facilities. Individuals living in the vicinity of large smelters and other industrial emitters of arsenic may be exposed to above-average arsenic levels both in the air, and as a result of atmospheric deposition, in water and soil and subsequent uptake into crops.

People sawing or drilling arsenic-treated wood without protective masks or burning this wood may be exposed to elevated levels of arsenic in air.

Recreational and subsistence fishers who consume appreciably higher amounts of locally caught fish from contaminated bodies of water may be exposed to higher levels of arsenic associated with dietary intake. Arsenic contamination has triggered the issuance of several human health advisories (EPA 1998g). As of December 1997, arsenic was identified as the causative pollutant in a restricted consumption advisory for the general population for all fish in a 7-mile area including Devil's Swamp Lake and Bayou Baton Rouge in Louisiana. A public health advisory has been issued for consumption of fish and shellfish from the Duwamish River, Seattle, Washington due to arsenic and other chemicals (WSDOE 2005).

6.8 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA, as amended, directs the Administrator of ATSDR (in consultation with the Administrator of EPA and agencies and programs of the Public Health Service) to assess whether adequate information on the health effects of arsenic is available. Where adequate information is not available, ATSDR, in conjunction with NTP, is required to assure the initiation of a program of research designed to determine the health effects (and techniques for developing methods to determine such health effects) of arsenic.

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The following categories of possible data needs have been identified by a joint team of scientists from ATSDR, NTP, and EPA. They are defined as substance-specific informational needs that if met would reduce the uncertainties of human health assessment. This definition should not be interpreted to mean that all data needs discussed in this section must be filled. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

6.8.1 Identification of Data Needs

Physical and Chemical Properties. The chemical and physical properties of the arsenic species of chief toxicological and environmental concern are sufficiently well characterized to allow estimation of the environmental fates of these compounds. However, more information regarding the K_{ow} and K_{oc} values of the organic arsenicals would help predict the fate of these compounds in the environment.

Production, Import/Export, Use, Release, and Disposal. According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit substance release and off-site transfer information to the EPA. The TRI, which contains this information for 2004, became available in May of 2006. This database is updated yearly and should provide a list of industrial production facilities and emissions.

While arsenic has not been produced in the United States since 1985, the United States is the largest consumer of arsenic and substantial quantities of arsenic are imported, primarily as arsenic trioxide (USGS 2006a). The agricultural use of inorganic arsenic pesticides have been discontinued in the United States. However, some organic arsenicals still may be used in agriculture. Current production and use data for individual arsenical pesticides and other arsenic compounds would help to estimate human exposure to the various arsenic species. Because arsenical pesticides are so persistent, a more complete picture of past use of these products would enable us to predict what areas may contain high levels of arsenic in soil.

Comprehensive estimates on emissions of arsenic date to the early 1980s (Nriagu and Pacyna 1988). The industrial picture has changed considerably since then and emission controls are being mandated more and more. For example, emission factors for Canadian smelters calculated in 1993 were grossly lower than those estimated in 1983 (Skeaff and Dubreuil 1997). There is a need for accurate and up-to-date measurements of atmospheric arsenic releases from both natural and anthropogenic sources to better assess human exposure to arsenic and guide environmental protection measures.

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Environmental Fate. The interconversion of the various arsenic species and transport among the environmental media is complex and not all aspects are well-studied. Additional quantitative data on the rates of oxidation, reduction, and biotransformation reactions of arsenic compounds, and how these depend on environmental conditions would be useful in evaluating and predicting the fate and transport of arsenic at hazardous waste sites and other areas.

Bioavailability from Environmental Media. Toxicokinetic and toxicity studies establish that bioaccessible (e.g., soluble, not strongly adsorbed to soil or embedded in minerals) arsenic is highly absorbed following inhalation and oral exposure (see Sections 3.4.1.2 and 3.4.1.1). Some work has been done on the effect of environmental matrix (soil, food) on accessibility and absorption of arsenic (Davis et al. 1992, 1996; Hamel et al. 1998; Roberts et al. 2002, 2007), but additional data would be valuable. Limited data suggests that dermal absorption of arsenic is very low (see Section 3.4.1.3) (Lowney et al. 2005), further data would be useful to establish whether arsenic uptake occurs from contact with contaminated soil or water, since humans may be exposed by these routes near hazardous waste sites.

Food Chain Bioaccumulation. Bioconcentration factors have been measured for several freshwater and marine species. While some species (mainly marine algae and shellfish) tend to bioconcentrate arsenic (EPA 1980a; Roper et al. 1996), it is not biomagnified through the food chain (Eisler 1994; EPA 1979, 1982b, 1983e, 2003b; Williams et al. 2006).

Carrots growing on land containing somewhat more than the permissible of arsenic in crop land did not contain levels of arsenic that were harmful (Helgesen and Larsen 1998). However, further research on the uptake of arsenic by a variety of plants in a wide range of arsenic polluted sites (e.g., mining area, orchards previously treated with lead arsenate) would be valuable in assessing human exposure near such sites through the consumption of vegetables from home gardens.

Exposure Levels in Environmental Media. Additional reliable monitoring data for the levels of arsenic in contaminated media at hazardous waste sites are needed or need to be made available, so that the information obtained on levels of arsenic in the environment can be used in combination with the known body burden of arsenic to assess the potential risk of adverse health effects in populations living in the vicinity of hazardous waste sites.

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Extensive monitoring data are available for total arsenic in all environmental media. Additional monitoring studies, specifically those that include identification of arsenic species, would allow more precise estimation of current exposure levels and possible human health risks.

Exposure Levels in Humans. Arsenic has been detected in human tissues, including blood, urine, hair, nails, and internal organs. Data are available for populations exposed in the workplace and for the general population (de Peyster and Silvers 1995; Jensen and Olsen 1995; Nygren et al. 1992), and some studies have been published on exposures near waste sites (Hwang et al. 1997a; Tsuji et al. 2005). Additional biomonitoring studies of residents near waste sites that contain arsenic would be helpful in evaluating the likely human health risks from these sites.

While some data are available on the speciation of arsenic in food, additional data on the particular species of arsenic, rather than just the total arsenic concentration, present in foods, especially seafood, are needed to better estimate the potential hazards to human health by the consumption of these foods (Ryan et al. 2001).

This information is necessary for assessing the need to conduct health studies on these populations.

Exposures of Children. Contaminated soils pose a particular hazard to children because of pica and hand-to-mouth activities. Some studies have been performed on exposure and body burden (Hwang et al. 1997a), but additional studies, including investigations of unique pathways for exposures of children and the amount of soil a child ingests, would provide valuable data. Small amounts of arsenic were found to be transferred to hands of children playing on play structures constructed from CCA-treated wood (Hemond and Solo-Gabriele 2004; Kwon et al. 2004; Shalat et al. 2006; Ursitti et al. 2004; Wang et al. 2005). Based on a review of existing studies, Hemond and Solo-Gabriele (2004) estimated that children with contact with CCA-treated wood may be subjected to doses in the range of tens of micrograms of arsenic per day and suggested that exposure by this route warrants further study. The PTDI assigned by the Food and Agriculture Organization of the United Nations and the World Health Organization (FAO/WHO) applies to adults. Studies are needed to assess whether children are different in their weight adjusted intake of arsenic. No childhood-specific means for reducing exposure were identified.

Child health data needs relating to susceptibility are discussed in Section 3.12.2, Identification of Data Needs: Children's Susceptibility.

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Exposure Registries. No exposure registries for arsenic were located. This substance is not currently one of the compounds for which a sub-registry has been established in the National Exposure Registry. The substance will be considered in the future when chemical selection is made for sub-registries to be established. The information that is amassed in the National Exposure Registry facilitates the epidemiological research needed to assess adverse health outcomes that may be related to exposure to this substance.

6.8.2 Ongoing Studies

The Federal Research in Progress (FEDRIP 2006) database provides additional information obtainable from a few ongoing studies that may fill in some of the data needs identified in Section 6.8.1. These studies are summarized in Table 6-7.

The U.S. Geological Survey, along with other federal and state agencies, industry, and academia, is conducting the National Geochemical Survey (NGS) in order to produce a body of geochemical data for the United States based primarily on stream sediments that have been analyzed using a consistent set of analytical methods. The goal of the NGS is to analyze at least one stream sediment sample in every 289 km² area by a single analytical method across the entire United States (USGS 2007b).

EPA is conducting a 4-year (2000–2003) national screening-level study of contaminants in freshwater fish, referred to as the National Fish Tissue Study (EPA 2004c). This study will allow the EPA to develop national estimates of the mean concentrations of 268 chemicals in tissues of fish from lakes and reservoirs of the coterminous United States. EPA analysis of the data from this study was scheduled to begin in January 2005, with the final report scheduled to be released in 2006. Interim raw data have been released each year, and are available from EPA. Fish samples have been analyzed for total inorganic arsenic (As(III) and As(V) combined), arsenic(III), arsenic(V), MMA(V), and DMA(V). Analysis for total arsenic was not performed as part of this study.

The American Water Works Association Research Foundation (AWWARF) supports research on arsenic in drinking water.

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Table 6-7. Ongoing Studies on the Environmental Fate and Exposure of Humans to Arsenic

Investigator	Affiliation	Research description	Sponsor
Basta, NT	Ohio State University, School of Natural Resources, Columbus, Ohio	Heavy metal and trace element biogeochemistry in soils; chemical speciation, bioavailability, and toxicity	USDA
Blum, CB	Columbia University Health Sciences, New York, New York	Bioavailability lead and arsenic in soil to humans	NIEHS
Hamilton, JW	Dartmouth College, Hanover, New Hampshire	Toxic metals—biological and environmental implications	NIEHS
Hoppin, J	Not specified	Monitoring of arsenic and other compounds in the blood and urine of a cohort of pregnant women in Norway	NIEHS
Kpomblekou, AK; Ankumah, RO	Tuskegee University, Agriculture and Home Economics, Tuskegee, Alabama	Biochemical processes in soils treated with trace-element-enriched broiler litter; to determine total arsenic and other metal concentrations and the distribution of their chemical forms in soils under long-term broiler litter treatments	USDA
Loeppert, RH	Texas A&M University, Soil and Crop Sciences, College Station, Texas	Inorganic chemical processes influencing soil and water quality	USDA
Peryea, FJ	Wenatchee Tree Fruit Research & Extension Center Washington State University, Pullman, Washington	Quantification of biogeochemical processes in lead arsenate-contaminated orchard soils and development of soil and plant management practices to minimize the toxicity risks that these soils impose on agricultural crops and to human and environmental health	USDA
Miller, DM; DeLaune, P; Miller, WP	University of Arkansas, Crop, Soil and Environmental Sciences, Fayetteville, Arkansas	Arsenic levels in soils of northwest Arkansas	USDA
van Geen, A	Columbia University, Lamont-Doherty Earth Observatory, Palisades, New York	Studies on arsenic in groundwater in Bangladesh	NSF

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Table 6-7. Ongoing Studies on the Environmental Fate and Exposure of Humans to Arsenic

Investigator	Affiliation	Research description	Sponsor
Walker, MJ et al.	University of Nevada, Natural Resources and Environmental Sciences, Reno, Nevada	Arsenic in Churchill County, Nevada domestic water supplies	USDA
Zheng, Y	Columbia University Health Sciences, New York, New York	Arsenic mobilization in Bangladesh groundwater	NIEHS

NIEHS = National Institute of Environmental Health Sciences; NSF = National Science Foundation; USDA = U.S. Department of Agriculture

Source: FEDRIP 2006