

6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

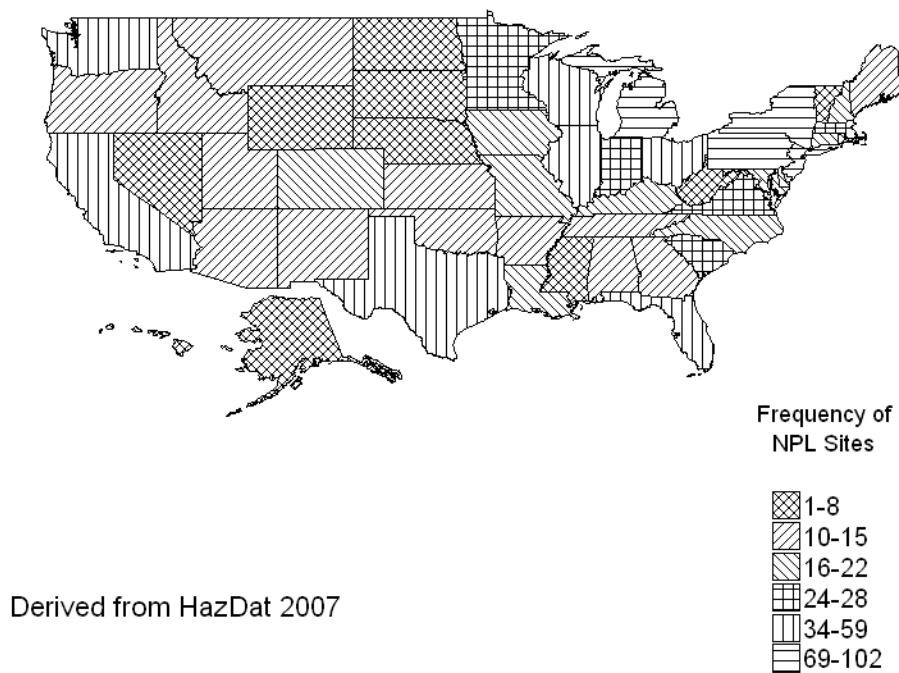
Chromium has been identified in at least 1,127 of the 1,699 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2007). However, the number of sites evaluated for chromium is not known. The frequency of these sites can be seen in Figure 6-1. Of these sites, 1,117 are located within the United States and 10 are located in the Commonwealth of Puerto Rico (not shown).

Human exposure to chromium occurs from both natural and anthropogenic sources. Chromium is present in the Earth's crust, with the main natural source of exposure being continental dust present in the environment (Barnhart 1997; Fishbein 1981; Pellerin and Booker 2000). Chromium is released into the environment in larger amounts as a result of human activities, which account for 60–70% of the total emissions of atmospheric chromium (Alimonti et al. 2000; Barceloux 1999; Seigneur and Constantinos 1995). This is indicated by the value of the enrichment factor (the enrichment factor relates the amount of chromium relative to an aluminum standard) of 3.5–8.1 (Dasch and Wolff 1989; Milford and Davidson 1985). Elements with enrichment factors >1 are assumed to have originated from anthropogenic sources (Schroeder et al. 1987). Of the estimated 2,700–2,900 tons of chromium emitted to the atmosphere annually from anthropogenic sources in the United States, approximately one-third is in the hexavalent form (EPA 1990b; Johnson et al. 2006). Industrial releases to the air, water, and soil are also potential sources of chromium exposure, and account for the majority of the anthropogenic releases (Johnson et al. 2006). The electroplating, leather tanning, and textile industries release large amounts of chromium to surface waters (Avudainayagam et al. 2003; Fishbein 1981; Johnson et al. 2006). Disposal of chromium-containing commercial products and coal ash from electric utilities and other industries are major sources of chromium releases into the soil (Barceloux 1999; Nriagu and Pacyna 1988). Solid waste and slag produced during chromate manufacturing processes when disposed of improperly in landfills can be potential sources of chromium exposure as well (Barceloux 1999; Kimbrough et al. 1999).

Chromium is primarily removed from the atmosphere by fallout and precipitation. The residence time of chromium in the atmosphere has not been directly measured, but by using copper as a model, it is expected to be <10 days (Nriagu 1979). The arithmetic mean concentrations of total chromium in the ambient air in United States, urban, suburban, and rural areas monitored during 1977–1984 ranged from 5 to 525 ng/m³, with the vast majority of samples <100 ng/m³ (EPA 1984a, 1990b). Ambient air in the United States usually contains very little chromium; at most measuring stations, the concentration was

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



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<300 ng/m³ and median levels were <20 ng/m³. As a result of smoking, indoor air contaminated with chromium can be 10–400 times greater than outdoor air concentrations (WHO 2003).

Chromium in the aquatic phase occurs in the soluble state or as suspended solids adsorbed onto clayish materials, organics, or iron oxides. Most of the soluble chromium is present as chromium(VI) or as soluble chromium(III) complexes and generally accounts for a small percentage of the total. Soluble chromium(VI) may persist in some bodies of water, but will eventually be reduced to chromium(III) by organic matter or other reducing agents in water (Cary 1982; EPA 1984a; Lide 1998). The residence times of chromium (total) in lake water range from 4.6 to 18 years, with the majority of the chromium in lakes and rivers ultimately deposited in the sediments (Schmidt and Andren 1984). In the United States, chromium concentrations are up to 84 µg/L in surface water and 0.2–1 µg/L in rainwater (WHO 2003). Most drinking water supplies in the United States contain <5 µg/L of chromium (WHO 2003). In ocean water, the mean chromium concentration is 0.3 µg/L (Cary 1982). In the United States, the groundwater concentration of chromium is generally low, with measurements in the range of 2–10 µg/L in shallow groundwater; levels as high as 50 µg/L have been reported in some supplies (WHO 2003).

Total chromium concentrations in U.S. soils range from 1 to 2,000 mg/kg, with a mean of 37.0 mg/kg (USGS 1984). Chromium(III) in soil is mostly present as insoluble carbonate and oxide of chromium(III); therefore, it will not be mobile in soil. The solubility of chromium(III) in soil and its mobility may increase due to the formation of soluble complexes with organic matter in soil, with a lower soil pH potentially facilitating complexation (Avudainayagam et al. 2003). Chromium has a low mobility for translocation from roots to the aboveground parts of plants (Calder 1988; Cary 1982; EPA 1984a, 1985a; King 1988; Stackhouse and Benson 1989).

A common source of chromium exposure is from food. Total chromium levels in most foods typically range from <10 to 1,300 µg/kg, with the highest concentrations being found in meat, fish, fruits, and vegetables (WHO 2003). The general population is exposed to chromium by inhaling air, drinking water, or eating food or food supplements that contain chromium. However, the primary source of exposure for the general population and non-occupationally exposed workers to chromium comes from food sources, although drinking water can contribute significantly when the levels are >25 µg/L (WHO 2003).

Dermal exposure to chromium may also occur during the use of consumer products that contain chromium, such as fertilizer, wood treated with copper dichromate or chromated copper arsenate, and leather tanned with chromic sulfate. In addition, people who reside in the vicinity of chromium waste

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disposal sites and chromium manufacturing and processing plants have a greater probability of elevated chromium exposure (Pellerin and Booker 2000).

Exposure to chromium for occupational groups can be two orders of magnitude higher than the exposure to the general population (Hemminki and Vainio 1984). Occupational exposure to chromium occurs mainly from chromate production, stainless steel production and welding, chrome plating, production of ferrochrome alloys, chrome pigment production and user industries, and from working in tanning industries (Pellerin and Booker 2000; Stern 1982)

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 2005). 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 2005).

6.2.1 Air

Estimated releases of 512,134 pounds of chromium to the atmosphere from 2,011 domestic manufacturing and processing facilities in 2009, accounted for about 6.6% of the estimated total environmental releases from facilities required to report to the TRI (TRI09 2011). Estimated releases of 269,599 pounds of chromium compounds to the atmosphere from 1,334 domestic manufacturing and processing facilities in 2009, accounted for about 1% of the estimated total environmental releases from facilities required to report to the TRI (TRI09 2011). These releases are summarized in Tables 6-1 and 6-2.

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

State ^c	RF ^d	Reported amounts released in pounds per year ^b							
		Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	Total release		On- and off-site
							On-site ^j	Off-site ^k	
AL	45	3,708	565	0	32,551	37,542	7,720	66,646	74,366
AK	1	0	0	25,000	27,000	0	52,000	0	52,000
AZ	20	1,055	91	0	345,591	17,749	334,967	29,520	364,487
AR	25	5,067	280	0	4,070	250	5,078	4,589	9,667
CA	85	1,653	1,287	382	11,059	11,749	1,674	24,457	26,131
CO	21	290	29	0	28,935	27,412	27,627	29,039	56,666
CT	45	542	2,060	0	15,242	36,845	10,913	43,777	54,689
DE	2	5	0	0	334	0	5	334	339
FL	29	90	31	0	255,926	1,889	188,645	69,290	257,935
GA	44	1,003	396	0	47,228	1,886	1,320	49,193	50,513
ID	5	105	0	0	130,286	487	107,517	23,361	130,879
IL	109	4,982	26,157	0	24,175	17,870	5,012	68,172	73,184
IN	119	15,813	3,811	0	1,490,024	85,846	29,247	1,566,247	1,595,493
IA	43	2,183	65	0	61,992	109,972	2,245	171,968	174,213
KS	40	8,066	217	0	3,330	270	8,088	3,795	11,883
KY	49	7,123	623	0	22,447	9,475	7,222	32,446	39,668
LA	35	1,367	232	0	49,357	164	17,636	33,484	51,121
ME	8	88	122	0	43	2,970	88	3,134	3,222
MD	7	1	14	0	9	11	1	34	35
MA	34	1,026	350	0	12,580	10,706	1,281	23,380	24,662
MI	78	7,673	2,382	0	25,049	4,951	8,729	31,326	40,055
MN	38	2,163	39	0	7,146	69	2,163	7,255	9,418
MS	22	2,806	12	0	454	72	2,808	536	3,344
MO	37	298,980	513	0	4,206	76	298,995	4,781	303,775
MT	1	0	0	0	520,172	0	520,172	0	520,172
NE	11	809	257	0	82,754	75	46,541	37,354	83,895
NV	14	238	6,631	0	1,080,422	407	1,076,822	10,876	1,087,699
NH	15	126	45	0	26	1,125	126	1,195	1,321
NJ	21	440	17	0	26,248	141	448	26,398	26,846
NM	3	52	0	0	54	0	106	0	106
NY	49	4,612	6,544	0	48,786	880	19,512	41,309	60,821
NC	52	928	312	0	4,008	6,969	976	11,241	12,217
ND	5	21	9	0	10,453	0	10,236	247	10,483
OH	164	9,119	5,675	0	204,658	73,201	43,141	249,513	292,654
OK	66	2,389	59	0	6,561	1,551	2,394	8,167	10,560

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

State ^c	RF ^d	Reported amounts released in pounds per year ^b							
		Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	Total release		
							On-site ^j	Off-site ^k	On- and off-site
OR	14	2,726	43	0	73,071	24	70,154	5,710	75,864
PA	180	12,514	1,378	0	658,699	217,491	47,732	842,350	890,082
PR	5	29	0	0	20	0	29	20	49
RI	5	0	0	0	0	1	0	1	1
SC	51	6,009	2,892	0	99,752	868	6,103	103,417	109,520
SD	8	45	1	0	42,240	1	42,162	125	42,287
TN	49	1,629	837	0	57,445	13,005	1,900	71,017	72,917
TX	140	6,499	8,912	213,649	20,672	5,244	221,563	33,414	254,976
UT	18	624	10	0	27,087	3,726	19,141	12,308	31,448
VT	4	22	13	0	32	215	27	255	282
VA	22	172	20,799	0	49,635	396	19,270	51,733	71,002
WA	21	1,608	142	0	48,867	4,633	2,864	52,386	55,249
WV	7	2,984	706	0	14,833	501	2,991	16,034	19,024
WI	142	92,652	4,452	0	476,398	27,955	93,201	508,256	601,456
WY	3	95	2	0	41,823	176	41,892	204	42,096
Total	2,011	512,134	99,011	239,031	6,193,750	736,849	3,410,482	4,370,293	7,780,775

^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 onsite 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: TRI09 2011 (Data are from 2009)

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

State ^c	RF ^d	Reported amounts released in pounds per year ^b							
		Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	Total release		
							On-site ^j	Off-site ^k	On- and off-site
AK	4	192	10	0	1,051,055	0	1,051,257	0	1,051,257
AL	41	12,995	3,368	0	855,643	33,148	756,162	148,992	905,154
AR	25	1,830	209	0	89,381	51,460	61,330	81,549	142,880
AZ	19	1,876	42	0	1,462,139	1,739	1,460,380	5,416	1,465,796
CA	42	699	17,437	0	252,991	58,991	190,150	139,969	330,119
CO	11	510	15	0	394,416	250	346,695	48,496	395,191
CT	13	1,305	820	0	16,344	20,567	1,328	37,707	39,035
DE	4	208	297	0	508,021	0	27,157	481,369	508,526
FL	25	4,306	357	0	385,025	22,394	346,871	65,211	412,082
GA	41	6,050	3,428	0	560,164	4,984	549,328	25,298	574,625
HI	1	0	0	0	0	0			
IA	15	7,476	50	0	146,295	110	74,254	79,676	153,930
ID	1	5,922	4	0	464,262	5	470,188	5	470,194
IL	72	5,913	19,980	636	949,161	164,644	219,738	920,596	1,140,334
IN	76	17,409	196,624	900	2,128,495	693,344	1,490,246	1,546,525	3,036,771
KS	23	6,121	2,902	250	368,352	213,474	106,741	484,358	591,099
KY	45	21,332	9,012	0	2,670,506	44,468	972,602	1,772,715	2,745,318
LA	21	1,229	440	1	260,204	16,688	214,862	63,700	278,562
MA	11	538	6	0	28,267	15,101	617	43,295	43,911
MD	21	2,401	532	0	110,310	144,631	74,943	182,931	257,873
ME	6	133	450	0	31,822	71,378	543	103,240	103,783
MI	73	10,157	9,079	4,400	557,893	16,191	340,219	257,502	597,720
MN	25	1,573	14,278	0	127,028	15,383	75,650	82,612	158,261
MO	26	4,537	618	0	336,752	37,987	104,979	274,916	379,894
MS	33	3,277	1,026	2,297,309	1,008,735	32,026	3,168,752	173,623	3,342,374
MT	8	1,805	0	0	250,394	645	102,190	150,653	252,843
NC	42	7,004	39,737	0	1,299,616	16,645	1,183,540	179,461	1,363,001
ND	7	4,869	242	0	289,934	1,533	134,352	162,226	296,578
NE	11	3,629	2,770	0	84,154	5,697	75,712	20,538	96,251
NH	3	69	10	0	10,650	250	2,079	8,900	10,979
NJ	17	1,590	39,946	0	3,865	52,021	3,765	93,658	97,423
NM	4	659	0	0	180,656	0	181,315	0	181,315
NV	7	255	0	0	1,531,094	0	1,531,349	0	1,531,349
NY	21	898	1,103	0	64,214	1,238	32,491	34,962	67,453
OH	101	28,059	17,971	444,822	2,125,297	247,620	1,578,830	1,284,938	2,863,768

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

State ^c	RF ^d	Reported amounts released in pounds per year ^b							
		Air ^e	Water ^f	UI ^g	Land ^h	Other ⁱ	Total release		
							On-site ^j	Off-site ^k	On- and off-site
OK	19	3,389	306	0	465,657	3,844	444,881	28,315	473,196
OR	11	235	106	0	58,134	653	7,961	51,167	59,128
PA	113	50,601	15,177	0	2,004,874	235,591	818,001	1,488,242	2,306,243
PR	7	5,285	51	0	3,193	198	5,285	3,443	8,728
RI	5	0	0	0	300	0	0	300	300
SC	2	3,260	2,945	0	389,283	130,685	278,206	247,966	526,172
SD	27	13	8	0	0	0	13	8	20
TN	3	4,604	30,074	0	2,109,366	5,918	1,570,982	578,980	2,149,962
TX	43	16,668	3,020	31,886	1,316,207	110,630	1,283,443	194,969	1,478,412
UT	92	2,088	1,262	0	1,632,406	14,465	1,538,140	112,081	1,650,221
VA	15	1,002	6,116	0	171,669	359	163,200	15,947	179,146
WA	16	568	36,434	0	75,005	106,290	43,057	175,240	218,298
WI	23	6,195	4,745	0	204,646	68,916	11,886	272,616	284,502
WV	16	4,465	2,878	0	911,062	29,786	603,016	345,175	948,191
WY	47	4,399	180	0	211,926	636	190,665	26,476	217,141
Total	1,334	269,599	486,063	2,780,204	30,156,862	2,692,585	23,889,351	12,495,962	36,385,312

^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 onsite 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: TRI09 2011 (Data are from 2009)

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Total chromium has been identified in air samples at 48 of 1,699 current or former NPL hazardous waste sites where it was detected in some environmental media (HazDat 2007).

Continental dust flux is the main natural source of chromium in the atmosphere; volcanic dust and gas flux are minor natural sources of chromium in the atmosphere (Fishbein 1981). Chromium is released into the atmosphere mainly by anthropogenic stationary point sources, including industrial, commercial, and residential fuel combustion, *via* the combustion of natural gas, oil, and coal (Kimbrough et al. 1999; Pacyn and Pacyn 2001; Seigneur and Constantinous 1995). Other important anthropogenic stationary point sources of chromium emission to the atmosphere are metal industries, such as chrome plating and steel production (EPA 1990b; Johnson et al. 2006; Pacyn and Pacyn 2001). Approximately one-third of the atmospheric releases of chromium are believed to be in the hexavalent form, chromium(VI) (Johnson et al. 2006). Other potentially small sources of atmospheric chromium emission are cement-producing plants (cement contains chromium), the wearing down of asbestos brake linings that contain chromium, incineration of municipal refuse and sewage sludge, and emission from chromium-based automotive catalytic converters. Emissions from cooling towers that previously used chromate chemicals as rust inhibitors are also atmospheric sources of chromium (EPA 1984b, 1990b; Fishbein 1981).

6.2.2 Water

Estimated releases of 99,011 pounds of chromium to surface water from 2,011 domestic manufacturing and processing facilities in 2009, accounted for about 1% of the estimated total environmental releases from facilities required to report to the TRI (TRI09 2011). Estimated releases of 486,063 pounds of chromium compounds to surface water from 1,334 domestic manufacturing and processing facilities in 2009, accounted for about 1% of the estimated total environmental releases from facilities required to report to the TRI (TRI09 2011).

Total chromium has been identified in surface water and groundwater samples at 427 of 1,699 and 813 of 1,699 current or former NPL hazardous waste sites where it was detected in some environmental media (HazDat 2007).

On a worldwide basis, the major chromium source in aquatic ecosystems is domestic waste water effluents (32.2% of the total) (Barceloux 1999). The other major sources are metal manufacturing (25.6%), ocean dumping of sewage (13.2%), chemical manufacturing (9.3%), smelting and refining of nonferrous metals (8.1%), and atmospheric fallout (6.4%) (Nriagu and Pacyna 1988). Annual

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anthropogenic input of chromium into water has been estimated to exceed anthropogenic input into the atmosphere (Nriagu and Pacyna 1988). However, land erosion, a natural source of chromium in water, was not included in the Nriagu and Pacyna (1988) estimation of chromium contributions to the aquatic environment.

6.2.3 Soil

Estimated releases of 6,193,750 pounds of chromium to soils from 2,011 domestic manufacturing and processing facilities in 2009, accounted for about 80% of the estimated total environmental releases from facilities required to report to the TRI (TRI09 2011). An additional 239,031 pounds of chromium were released via underground injection. Estimated releases of 30,156,862 pounds of chromium compounds to soils from 1,334 domestic manufacturing and processing facilities in 2009, accounted for about 83% of the estimated total environmental releases from facilities required to report to the TRI (TRI09 2011). An additional 2,780,204 pounds were released via underground injection. These releases are summarized in Tables 6-1 and 6-2.

Total chromium has been identified in soil and sediment samples at 696 of 1,699 and 471 of 1,699 current or former NPL hazardous waste sites where it was detected in some environmental media (HazDat 2007).

On a worldwide basis, the disposal of commercial products that contain chromium may be the largest contributor, accounting for 51% of the total chromium released to soil (Nriagu and Pacyna 1988). Other significant sources of chromium release into soil include the disposal of coal fly ash and bottom fly ash from electric utilities and other industries (33.1%), agricultural and food wastes (5.3%), animal wastes (3.9%), and atmospheric fallout (2.4%) (Nriagu and Pacyna 1988). Solid wastes from metal manufacturing constituted <0.2% to the overall chromium release in soil. However, the amount of chromium in sludge or residue that is disposed of in landfills by manufacturing and user industries that treat chromate wastes in ponds and lagoons is not included in the estimation by Nriagu and Pacyna (1988).

6.3 ENVIRONMENTAL FATE**6.3.1 Transport and Partitioning**

Chromium is present in the atmosphere primarily in particulate form; naturally occurring gaseous forms of chromium are rare (Cary 1982; Kimbrough et al. 1999; Seigneur and Constantinos 1995). The

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transport and partitioning of particulate matter in the atmosphere depends largely on particle size and density. Atmospheric particulate matter is deposited on land and water via wet and dry deposition. Wet, dry, and total deposition rates of chromium and several other trace metals in remote, rural and urban areas were summarized by Schroeder et al. (1987). Deposition rates tended to be highest in urban areas that had greater atmospheric levels of chromium as compared to rural and remote locations. The rates of wet and dry deposition are dependent upon several factors, including particle and aerosol size distribution (Kimbrough et al. 1999). The mass mean aerodynamic diameter (MMAD) of chromium aerosols or particulates emitted from several industrial sources are $\leq 10 \mu\text{m}$ and it has been estimated that chromium-containing particulates emitted from these industrial sources can remain airborne for 7–10 days and are subject to long-range transport (Kimbrough et al. 1999). Based on a troposphere to stratosphere turnover time of 30 years (EPA 1979), atmospheric particles with a residence time of < 10 days are not expected to transport from the troposphere to the stratosphere and there are no data in the reviewed literature indicating that chromium particles are transported from the troposphere to the stratosphere (Pacyna and Ottar 1985).

Since chromium compounds cannot volatilize from water, transport of chromium from water to the atmosphere is not likely, except by transport in windblown sea sprays. Most of the chromium released into water will ultimately be deposited in the sediment. A very small percentage of chromium in the water column is present in both soluble and insoluble forms. In the aquatic phase, chromium(III) occurs mostly as suspended solids adsorbed onto clayish materials, organics, or iron oxide (Fe_2O_3) present in water. Approximately 10.5–12.6% of chromium in the aquatic phase of the Amazon and Yukon Rivers was in solution, the rest being present in the suspended solid phase (Cary 1982; King 1988). The ratio of chromium in suspended solids to dissolved form in an organic-rich river in Brazil was 2.1 (Malm et al. 1988).

The bioconcentration factor (BCF) for chromium(VI) in rainbow trout (*Salmo gairdneri*) is 1. In bottom-feeder bivalves, such as the oyster (*Crassostrea virginica*), blue mussel (*Mytilus edulis*), and soft shell clam (*Mya arenaria*), the BCF values for chromium(III) and chromium(VI) range from 86 to 192 (EPA 1980, 1984a; Fishbein 1981; Schmidt and Andren 1984). The bioavailability of chromium(III) to freshwater invertebrates (*Daphnia pulex*) decreased with the addition of humic acid (Ramelow et al. 1989). This decrease in bioavailability was attributed to lower availability of the free form of the metal due to its complexation with humic acid. Based on this information, chromium is not expected to biomagnify in the aquatic food chain. Although higher concentrations of chromium have been reported in plants growing in high chromium-containing soils (e.g., soil near ore deposits or chromium-emitting

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industries and soil fertilized by sewage sludge) compared with plants growing in normal soils, most of the increased uptake in plants is retained in roots, and only a small fraction is translocated in the aboveground part of edible plants (Cary 1982; WHO 1988). Therefore, bioaccumulation of chromium from soil to aboveground parts of plants is unlikely (Petruzzelli et al. 1987). There is no indication of biomagnification of chromium along the terrestrial food chain (soil-plant-animal) (Cary 1982).

The mobility of chromium in soil is dependent upon the speciation of chromium, which is a function of redox potential and the pH of the soil. In most soils, chromium will be present predominantly in the chromium(III) oxidation state. This form has very low solubility and low reactivity, resulting in low mobility in the environment (Barnhart 1997; Jardine et al. 1999; Robson 2003). Under oxidizing conditions, chromium(VI) may be present in soil as CrO_4^{2-} and HCrO_4^- (James et al. 1997). In this form, chromium is relatively soluble and mobile. A leachability study comparing the mobility of several metals, including chromium, in soil demonstrated that chromium had the least mobility of all of the metals studied (Sahuquillo et al. 2003). These results support previous data finding that chromium is not very mobile in soil, especially in the trivalent oxidation state (Balasoiu et al. 2001; Jardine et al. 1999; Lin et al. 1996; Robson 2003). These results are further supported by a leachability investigation in which chromium mobility was studied for a period of 4 years in a sandy loam (Sheppard and Thibault 1991). The vertical migration pattern of chromium in this soil indicated that after an initial period of mobility, chromium forms insoluble complexes and little leaching is observed. Chromium present as insoluble oxide, $\text{Cr}_2\text{O}_3 \cdot n\text{H}_2\text{O}$, exhibited limited mobility in soil (Rifkin et al. 2004). Flooding of soils and the subsequent anaerobic decomposition of plant detritus matters may increase the mobilization of chromium(III) in soils due to formation of soluble complexes (Stackhouse and Benson 1989). This complexation may be facilitated by a lower soil pH.

A smaller percentage of total chromium in soil exists as soluble chromium(VI) and chromium(III) complexes, which are more mobile in soil. Chromium that is irreversibly sorbed onto soil (e.g., in the interstitial lattice of goethite, FeOOH) will not be bioavailable to plants and animals under any condition. Organic matter in soil is expected to convert soluble chromate, chromium(VI), to insoluble chromium(III) oxide, Cr_2O_3 (Calder 1988). Surface runoff from soil can transport both soluble and bulk precipitate of chromium to surface water. Soluble and unadsorbed chromium(VI) and chromium(III) complexes in soil may leach into groundwater. The leachability of chromium(VI) in the soil increases as the pH of the soil increases. On the other hand, lower pH present in acid rain may facilitate leaching of acid-soluble chromium(III) complexes and chromium(VI) compounds in soil. Chromium has a low mobility for translocation from roots to aboveground parts of plants (Cary 1982). However, depending on the

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geographical areas where the plants are grown, the concentration of chromium in aerial parts of certain plants may differ by a factor of 2–3 (Cary 1982).

6.3.2 Transformation and Degradation

6.3.2.1 Air

In the atmosphere, chromium(VI) may be reduced to chromium(III) at a significant rate by vanadium (V^{2+} , V^{3+} , and VO^{2+}), Fe^{2+} , HSO_3^- , and As^{3+} (EPA 1987b; Kimbrough et al. 1999). Conversely, chromium(III), if present as a salt other than Cr_2O_3 , may be oxidized to chromium(VI) in the atmosphere if 1% of the manganese in atmospheric aerosols is present as MnO_2 (EPA 1990b). The estimated atmospheric half-life for chromium(VI) reduction to chromium(III) was reported in the range of 16 hours to about 5 days (Kimbrough et al. 1999).

6.3.2.2 Water

The reduction of chromium(VI) to chromium(III) and the oxidation of chromium(III) to chromium(VI) in water has been investigated extensively. Reduction of chromium(VI) to chromium(III) can occur under suitable conditions in the aqueous environment, if an appropriate reducing agent is available. The most common reducing agents present in aqueous systems include: organic matter; hydrogen sulfide; sulfur, iron sulfide; ammonium; and nitrate (Kimbrough et al. 1999). The reduction of chromium(VI) by S^{2-} or Fe^{+2} ions under anaerobic conditions occurs rapidly, with the reduction half-life ranging from instantaneous to a few days (Seigneur and Constantinos 1995). However, the reduction of chromium(VI) by organic sediments and soils was much slower and depended on the type and amount of organic material and on the redox condition of the water. The reduction half-life of chromium(VI) in water with soil and sediment ranged from 4 to 140 days, with the reaction typically occurring faster under anaerobic rather than aerobic conditions (Saleh et al. 1989). Generally, the reduction of chromium(VI) to chromium(III) is also favored under acidic conditions (Kimbrough et al. 1999).

Oxidation of chromium(III) to chromium(VI) can also occur in the aqueous environment, depending on several factors. Although oxygen is known to oxidize chromium(III) to chromium(VI), dissolved oxygen by itself in natural waters did not cause any measurable oxidation of chromium(III) to chromium(VI) over a period of 128 days (Saleh et al. 1989). When chromium(III) was added to lake water, a slow oxidation of chromium(III) to chromium(VI) occurred, corresponding to an oxidation half-life of nine years. Addition of 50 mg/L manganese oxide accelerated the process, decreasing the oxidation half-life to

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approximately 2 years (Saleh et al. 1989). The oxidation of chromium(III) to chromium(VI) during chlorination of water was highest in the pH range of 5.5–6.0 (Saleh et al. 1989). However, the process would rarely occur during chlorination of drinking water because of the low concentrations of chromium(III) in these waters, and the presence of naturally occurring organics that may protect chromium(III) from oxidation, either by forming strong complexes with chromium(III) or by acting as a reducing agent to free available chlorine (EPA 1988c). In chromium(III)-contaminated waste waters having pH ranges of 5–7, chlorination may convert chromium(III) to chromium(VI) in the absence of chromium(III)-complexing and free chlorine reducing agents (EPA 1988c).

Chromium speciation in groundwater also depends on the redox potential and pH conditions in the aquifer. Chromium(VI) predominates under highly oxidizing conditions; whereas chromium(III) predominates under reducing conditions. Oxidizing conditions are generally found in shallow, oxygenated aquifers, and reducing conditions generally exist in deeper, anaerobic groundwaters. In natural groundwater, the pH is typically 6–8, and CrO_4^{-2} is the predominant species of chromium in the hexavalent oxidation state, while $\text{Cr}(\text{OH})_2^{+1}$ will be the dominant species in the trivalent oxidation state. This species and other chromium(III) species will predominate in more acidic pH; $\text{Cr}(\text{OH})_3$ and $\text{Cr}(\text{OH})_4^{-1}$ predominate in more alkaline waters (Calder 1988).

6.3.2.3 Sediment and Soil

The fate of chromium in soil is greatly dependent upon the speciation of chromium, which is a function of redox potential and the pH of the soil. In most soils, chromium will be present predominantly in the chromium(III) state. This form has very low solubility and low reactivity resulting in low mobility in the environment and low toxicity in living organisms (Ashley et al. 2003; Barnhart 1997; EPA 1994b).

Under oxidizing conditions, chromium(VI) may be present in soil as CrO_4^{-2} and HCrO_4^- (James et al. 1997). In this form, chromium is relatively soluble, mobile, and toxic to living organisms. In deeper soil where anaerobic conditions exist, chromium(VI) will be reduced to chromium(III) by S^{-2} and Fe^{+2} present in soil. The reduction of chromium(VI) to chromium(III) is possible in aerobic soils that contain appropriate organic energy sources to carry out the redox reaction, with the reduction of chromium(VI) to chromium(III) facilitated by low pH (Cary 1982; EPA 1990b; Saleh et al. 1989).

The oxidation of chromium(III) to chromium(VI) in soil is facilitated by the presence of organic substances, oxygen, manganese dioxide, moisture, and the elevated temperatures in surface soil that result from brush fires (Calder 1988; Cary 1982). Organic forms of chromium(III) (e.g., humic acid complexes)

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are more easily oxidized than insoluble oxides. However, oxidation of chromium(III) to chromium(VI) was not observed in soil under conditions of maximum aeration and a maximum pH of 7.3 (Bartlett and Kimble 1976). It was later reported that soluble chromium(III) in soil can be partly oxidized to chromium(VI) by manganese dioxide in soil, and the process is enhanced at pH values >6 (Bartlett 1991). Because most chromium(III) in soil is immobilized due to adsorption and complexation with soil materials, the barrier to this oxidation process is the lack of availability of mobile chromium(III) to immobile manganese dioxide in soil surfaces. Due to this lack of availability of mobile chromium(III) to manganese dioxide surfaces, a large portion of chromium in soil will not be oxidized to chromium(VI), even in the presence of manganese dioxide and favorable pH conditions (Bartlett 1991; James et al. 1997).

The microbial reduction of chromium(VI) to chromium(III) has been discussed as a possible remediation technique in heavily contaminated environmental media or wastes (Chen and Hao 1998; EPA 1994b). Factors affecting the microbial reduction of chromium(VI) to chromium(III) include biomass concentration, initial chromium(VI) concentration, temperature, pH, carbon source, oxidation-reduction potential, and the presence of both oxyanions and metal cations. Although high levels of chromium(VI) are toxic to most microbes, several resistant bacterial species have been identified that could ultimately be employed in remediation strategies (Chen and Hao 1998; EPA 1994b). Elemental iron, sodium sulfite, sodium hydrosulfite, sodium bisulfite, sodium metabisulfite sulfur dioxide, and certain organic compounds such as hydroquinone have also been shown to reduce chromium(VI) to chromium(III) and have been discussed as possible remediation techniques in heavily contaminated soils (Higgins et al. 1997; James et al. 1997). The limitations and efficacy of these and all remediation techniques are dependent upon the ease in which the reducing agents are incorporated into the contaminated soils.

6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

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

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6.4.1 Air

Levels of total chromium in the ambient air in U.S. urban and nonurban areas during 1977–1984 are reported in EPA's National Aerometric Data Bank (EPA 1984a, 1990b). According to this databank, the arithmetic mean total chromium concentrations from a total of 2,106 monitoring stations ranged from 5 to 525 ng/m³. The two locations that showed the highest total arithmetic mean chromium concentrations were in Steubenville, Ohio, in 1977 (525 ng/m³) and in Baltimore, Maryland, in 1980 (226 ng/m³) (EPA 1990b). Arithmetic mean total chromium concentrations in only 8 of 173 sites monitored in 1984 were >100 ng/m³ (EPA 1990b).

An indoor/outdoor air study was conducted in southwestern Ontario to measure levels of chromium(VI) and the size fraction of chromium(VI). Indoor and outdoor samples were taken from 57 homes during the summer months of 1993. The concentrations were 0.1–0.6 ng/m³ indoors (geometric mean 0.2 ng/m³) and were 0.10–1.6 ng/m³ outdoors (geometric mean 0.55 ng/m³). The indoor concentrations were less than half of the outdoor concentrations. Analysis of airborne chromium(VI) particles showed that they were inhalable in size (Bell and Hipfner 1997). A study measured the levels of chromium(VI) and total chromium in the ambient air in Hudson County, New Jersey. The concentrations of chromium(VI) in the indoor air of residences in Hudson County in 1990 ranged from 0.38 to 3,000 ng/m³, with a mean of 1.2 ng/m³ (Falerios et al. 1992).

Another study analyzed the relationship between soil levels of chromium and chromium content of the atmosphere. An indoor/outdoor study was conducted at 25 industrial sites in Hudson County, New Jersey to analyze soils containing chromite ore processing residues. The industrial sites include industrial, manufacturing, trucking, and warehouse facilities. The study found industrial indoor chromium(VI) and total chromium concentrations to be 0.23–11 and 4.1–130 ng/m³ and industrial outdoor chromium(VI) and total chromium concentrations to be 0.013–15.3 and 1.9–84.5 ng/m³. The results of this study found that higher levels of chromium(VI) in soil do not necessarily result in higher levels of chromium(VI) in air (Finley et al. 1993). The mean concentration of total chromium at the same sites was 7.1 ng/m³, with a concentration range of 3.7–12 ng/m³. Monitoring data in Hudson County, New Jersey has shown a background chromium(VI) concentration of 0.2–3.8 ng/m³ with a mean concentration of 1.2 ng/m³ (Scott et al. 1997a). The airborne total chromium concentration range was 1.5–10 ng/m³ with a mean concentration of 4.5 ng/m³ (Scott et al. 1997a). The mean airborne chromium(VI) and total chromium concentrations in the indoor air of industrial sites in Hudson County, New Jersey, contaminated by chromite ore-processing residue were 3 ng/m³ (range, 0.23–11 ng/m³) and 23 ng/m³ (range, 4.11–

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130 ng/m³), respectively. The mean airborne chromium(VI) and total chromium concentrations in outdoor air for the same sites were 9.9 ng/m³ (range, 0.13–110 ng/m³) and 37 ng/m³ (range, 1.9–250 ng/m³), respectively (Falerios et al. 1992).

An air dispersion model was developed which accurately estimated chromium(VI) concentrations at two of these industrial sites in Hudson County, New Jersey (Scott et al. 1997b). The background corrected airborne concentrations in ng/m³ for seven sampling dates are reported as measured (modeled values in parentheses): 0.0 (0.41); 6.2 (7.7); 0.9 (1.7); 2.8 (2.7); 0.0 (0.08); 0.3 (0.1); and 1.2 (0.12). The estimated percent levels of chromium(III) and chromium(VI) in the U.S. atmosphere from anthropogenic sources are given in Table 6-3 (EPA 1990b). Fly ash from a coal-fired power plant contained 1.4–6.1 mg/kg chromium(VI) (Stern et al. 1984). In a field study to assess inhalation exposure to chromium during showering and bathing activities, the average chromium(VI) concentration in airborne aerosols ranged from 87 to 324 ng/m³ when water concentrations of 0.89–11.5 mg/L of chromium(VI) were used in a standard house shower (Finley et al. 1996a).

The concentrations of atmospheric chromium in remote areas, such as the Arctic region and the South Pole, range from 0.005 to 2.6 ng/m³ (Barrie and Hoff 1985; Cary 1982; Schroeder et al. 1987; Sheridan and Zoller 1989). Saltzman et al. (1985) compared the levels of atmospheric chromium at 59 sites in U.S. cities during 1968–1971 with data from EPA's National Aerometric Data Bank file for 1975–1983. They concluded that atmospheric chromium levels may have declined in the early 1980s from the levels detected in the 1960s and 1970s.

6.4.2 Water

Chromium concentrations in U.S. river water usually range from <1 to 30 µg/L (EPA 1984a; Malm et al. 1988; Ramelow et al. 1987), with a median value of 10 µg/L (Eckel and Jacob 1988; Smith et al. 1987). Chromium concentrations in lake water generally do not exceed 5 µg/L (Borg 1987; Cary 1982), with higher levels of chromium related to anthropogenic pollution sources. Dissolved chromium concentrations of 0.57–1.30 µg/L were reported in the Delaware River near Marcus Hook and Fieldsboro, Pennsylvania in January 1992, with chromium(III) composing 67% of the total (Riedel and Sanders 1998). In March 1992, these concentrations decreased to 0.03–0.23 µg/L. In general, the concentration of chromium in ocean water is much lower than that in lakes and rivers. The mean chromium concentration in ocean water is 0.3 µg/L, with a range of 0.2–50 µg/L (Cary 1982). The mean concentration of chromium in rainwater is 0.14–0.9 µg/L (Barrie et al. 1987; Dasch and Wolff 1989).

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Table 6-3. Estimates of U.S. Atmospheric Chromium Emissions from Anthropogenic Sources

Source category	Estimated number of sources	Chromium emissions (metric tons/year)	Estimated hexavalent chromium (percent)
Combustion of coal and oil	Many	1,723	0.2
Chromium chemical manufacturing	2	18	67
Chemical manufacturing cooling towers	2,039	43	100
Petroleum refining cooling towers	475	32	100
Specialty/steel production	18	103	2.2
Primary metal cooling towers	224	8	100
Chrome plating	4,000	700	~100
Comfort cooling towers	38,000	7.2–206	100
Textile manufacturing cooling towers	51	0.1	100
Refractory production	10	24	1.3
Ferrochromium production	2	16	5.4
Sewage sludge incineration	133	13	<0.1
Tobacco cooling towers	16	0.2	100
Utility industry cooling towers	6	1.0	100
Chrome ore refining	6	4.8	<0.1
Tire and rubber cooling towers	40	0.2	100
Glass manufacturing cooling towers	3	0.01	100
Cement production	145	3	0.2
Municipal refuse incineration	95	2.5	0.3
National total		2,700–2,900	

Source: EPA 1990b

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The concentrations of total chromium in groundwater at the Idaho National Engineering Laboratory, where chromate is used as a corrosion inhibitor, ranged from <1 to 280 µg/L (USGS 1989). The water from a village well situated near a waste pond receiving chromate waste in Douglas, Michigan, contained 10,800 µg/L chromium(VI). Similarly, water from a private well adjacent to an aircraft plant in Nassau County, New York, contained 25,000 µg/L chromium(VI), while water from a public well adjacent to another aircraft plant in Bethpage, New York, contained 1,400 µg/L chromium(VI) (Davids and Lieber 1951). In a later study, water from an uncontaminated well in Nassau County, New York, contained an undetectable level of chromium(VI), whereas a contaminated well in the vicinity of a plating plant contained 6,000 µg/L chromium(VI) (Lieber et al. 1964). A chromium concentration of 120 µg/L was detected in private drinking water wells adjacent to an NPL site in Galena, Kansas (Agency for Toxic Substances and Disease Registry 1990a).

The chromium levels detected in drinking water in an earlier study (1962–1967 survey) may be erroneous due to questionable sampling and analytical methods (see Section 7.1) (EPA 1984a). Total chromium levels in drinking water were reported to range from 0.2 to 35 µg/L (EPA 1984a). Most drinking water supplies in the United States contain <5 µg/L of total chromium (WHO 2003). The concentration of chromium in household tap water may be higher than supply water due to corrosion of chromium-containing pipes. At a point of maximum contribution from corrosion of the plumbing system, the peak chromium in tap water in Boston, Massachusetts was 15 µg/L (Ohanian 1986). A survey that targeted drinking waters from 115 Canadian municipalities during 1976–1977 reported the median chromium concentration to be <2.0 µg/L (detection limit) and the maximum chromium concentration to be 8.0 µg/L (Meranger et al. 1979). A recent monitoring survey of drinking water by the California Department of Public Health found that levels of chromium(VI) were <10 µg/L in 86% (2,003 out of 2,317 sources) of the drinking water sources sampled; however, levels above 50 µg/L were noted in six sources (CDPH 2007). In this survey, a source was defined as those reporting more than a single detection of chromium(VI) and may include both raw and treated sources, distribution systems, blending reservoirs, and other sampled entities. These data did not include agricultural wells, monitoring wells, or more than one representation of the same source (e.g., a source with both raw and treated entries is counted a single source).

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6.4.3 Sediment and Soil

The chromium level in soils varies greatly and depends on the composition of the parent rock from which the soils were formed. Basalt and serpentine soils, ultramafic rocks, and phosphorites may contain chromium as high as a few thousand mg/kg (Merian 1984) whereas soils derived from granite or sandstone will have lower concentrations of chromium (Swaine and Mitchell 1960). The concentration range of chromium in 1,319 samples of soils and other surficial materials collected in the conterminous United States was 1–2,000 mg/kg, with a geometric mean of 37 mg/kg (USGS 1984). Chromium concentrations in Canadian soils ranged from 5 to 1,500 mg/kg, with a mean of 43 mg/kg (Cary 1982). In a study with different kinds of soils from 20 diverse sites including old chromite mining sites in Maryland, Pennsylvania, and Virginia, the chromium concentration ranged from 4.9 to 71 mg/kg (Beyer and Cromartie 1987). A polynuclear aromatic hydrocarbon (PAH) soil study was conducted to determine the metal levels in soil at the edge of a busy road that runs through the Aplerbecker Forest in West Germany. Chromium(VI) concentrations of 64 mg/kg were measured, and these concentrations were 2- to 4-fold higher along the road than in the natural forest (Munch 1993). The soil beneath decks treated with chrominated copper arsenate (CCA), a wood preservative, had an average chromium content of 43 mg/kg (Stilwell and Gorny 1997).

Chromium has been detected at a high concentration (43,000 mg/kg) in soil at the Butterworth Landfill site in Grand Rapid City, Michigan, which was a site listed on the NPL (Agency for Toxic Substances and Disease Registry 1990b).

Chromium was detected in sediment obtained from the coastal waters of the eastern U.S. seashore at concentrations of 3.8–130.9 µg/g in 1994 and 0.8–98.1 µg/g in 1995 (Hyland et al. 1998).

6.4.4 Other Environmental Media

The concentration of chromium in the particulate portion of melted snow collected from two urban areas (Toronto and Montreal) of Canada ranged from 100 to 3,500 mg/kg (Landsberger et al. 1983). In the suspended materials and sediment of water bodies, chromium levels ranged from 1 to 500 mg/kg (Byrne and DeLeon 1986; EPA 1984a; Mudroch et al. 1988; Ramelow et al. 1987). The chromium concentration in incinerated sewage sludge ash may be as high as 5,280 mg/kg (EPA 1984a).

Total chromium levels in most fresh foods are extremely low (vegetables [20–50 µg/kg], fruits [20 µg/kg], and grains and cereals [40 µg/kg]) (Fishbein 1984). The chromium levels of various foods are

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reported in Table 6-4. In a study to find the concentrations of chromium in edible vegetables in Tarragon Province, Spain, the highest levels of chromium were found in radish root and spinach, with a nonsignificant difference between the samples collected in two areas (northern industrial and southern agricultural). The samples ranged in concentration from 0.01 to 0.21 $\mu\text{g/g}$ (industrial) and from 0.01 to 0.22 $\mu\text{g/g}$ (agricultural) (Schuhmacker et al. 1993). Acidic foods that come into contact with stainless steel surfaces during harvesting, processing, or preparation for market are sometimes higher in chromium content because of leaching conditions. Processing, however, removes a large percentage of chromium from foods (e.g., whole-grain bread contains 1,750 $\mu\text{g/kg}$ chromium, but processed white bread contains only 140 $\mu\text{g/kg}$; and molasses contains 260 $\mu\text{g/kg}$ chromium, but refined sugar contains only 20 $\mu\text{g/kg}$ chromium) (Anderson 1981; EPA 1984a).

Chromium levels in oysters, mussels, clams, and mollusks vary from <0.1 to 6.8 mg/kg (dry weight) (Byrne and DeLeon 1986; Ramelow et al. 1989). Fish and shellfish collected from ocean dump sites off New York City, Delaware Bay, and New Haven, Connecticut, contained <0.3 –2.7 mg/kg chromium (wet weight) (Greig and Jones 1976). The chromium concentration in fish sampled from 167 lakes in the northeastern United States was 0.03–1.46 $\mu\text{g/g}$ with a mean concentration of 0.19 $\mu\text{g/g}$ (Yeardley et al. 1998). Higher levels of chromium in forage of meat animals have been reported for plants grown in soils with a high concentration of chromium (see Section 6.3.1). Cigarette tobacco reportedly contains 0.24–14.6 mg/kg chromium, but no estimates were available regarding the chromium levels in inhaled cigarette smoke (Langård and Norseth 1986). Cigarette tobacco grown in the United States contains ≤ 6.3 mg/kg chromium (IARC 1980).

Cement-producing plants are a potential source of atmospheric chromium. Portland cement contains 41.2 mg/kg chromium (range 27.5–60 mg/kg). Soluble chromium accounts for 4.1 mg/kg (range 1.6–8.8 mg/kg) of which 2.9 mg/kg (range 0.03–7.8 mg/kg) is chromium(VI) (Fishbein 1981). The wearing down of vehicular brake linings that contain asbestos represents another source of atmospheric chromium. Asbestos may contain $\approx 1,500$ mg/kg of chromium. The introduction of catalytic converters on U.S. automobiles in 1975 in the United States represented an additional source of atmospheric chromium. Catalysts, such as copper chromite, emit $<10^6$ metal-containing condensation nuclei per cubic centimeter in vehicular exhaust, under various operating conditions (Fishbein 1981).

Fertilizers also contain chromium and other metals. Commercially available bulk fertilizers had total chromium levels of 50–114 $\mu\text{g/g}$ (Gorman et al. 2011).

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Table 6-4. Chromium Content in Various U.S. Foods

Sample	Mean concentration (µg/kg)	Reference
Fresh vegetables	30–140	EPA 1984a
Frozen vegetables	230	EPA 1984a
Canned vegetables	230	EPA 1984a
Fresh fruits	90–190	EPA 1984a
Fruits	20	EPA 1984a
Canned fruits	510	EPA 1984a
Dairy products	100	EPA 1984a
Chicken eggs	60	Kirpatrick and Coffin 1975
Chicken eggs	160	Schroeder et al. 1962
Chicken eggs	520	Gormican 1970
Whole fish	50–80	EPA 1984a
Edible portion of fresh fin fish	<100–160	Eisenberg and Topping 1986
Meat and fish	110–230	EPA 1984a
Seafoods	120–470	EPA 1984a
Grains and cereals	40–220	EPA 1984a
Sugar, refined ^a	<20	WHO 1988

^aValue in Finnish sugar

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6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

The general population is exposed to chromium by inhaling ambient air, ingesting food, and drinking water containing chromium. Home-based exposures can occur to the families of occupational workers in what is known as worker-to-family exposures. Home exposures can also occur through proximity to hazardous waste sites (Pellerin and Booker 2000). A study measured the relationships between chromium in household dust and chromium in the urine of Hudson County, New Jersey residents (Pellerin and Booker 2000). Three major producers of chromium(VI) in the form of chromate were active in the area for over 70 years, and produced over 2 million tons of chromium containing slag waste (Pellerin and Booker 2000). Chromium(VI) levels as high as several hundred parts per million were measured in some of the soil samples extracted from the area (Pellerin and Booker 2000). Similarly, residents living in an area of Taiwan with a large number of electroplating industries had elevated blood chromium levels, and a significant correlation between blood chromium levels and soil chromium levels were found (Chiang et al. 2011).

Dermal exposure of the general public to chromium can occur from skin contact with consumer products that contain chromium. Some of the consumer products known to contain chromium are certain wood preservatives, cement, cleaning materials, textiles, and leather tanned with chromium (WHO 1988). Both chromium(III) and chromium(VI) are known to penetrate the skin, although chromium(VI) penetrates to a higher degree (Robson 2003). Elevated levels of chromium were observed in urine and blood of people who have had knee or hip replacements made with chromium containing metal alloys (Price 2011; Sunderman et al. 1989). Levels of chromium in ambient air ($<0.01\text{--}0.03\ \mu\text{g}/\text{m}^3$) (Fishbein 1984; Pellerin and Booker 2000) and tap water ($<1\ \mu\text{g}/\text{L}$) (Pellerin and Booker 2000) have been used to estimate the daily intake of chromium via inhalation ($<0.2\text{--}0.6\ \mu\text{g}$) and tap water ($<4\ \mu\text{g}$). These estimates are based on an air inhalation rate of $20\ \text{m}^3/\text{day}$ and a drinking water consumption rate of $2\ \text{L}/\text{day}$. The daily chromium intake for the U.S. population from consumption of selected diets (diets with 25 and 43% fat) has been estimated to range from 25 to 224 μg with an average of 76 μg (Kumpulainen et al. 1979). The chromium concentrations in tissues and body fluids of the general population are given in Table 6-5.

Workers in industries that use chromium can be exposed to concentrations of chromium two orders of magnitude higher than exposure to the general population, and workers in some 80 different professions may be exposed to chromium(VI) (Hemminki and Vainio 1984; Pellerin and Booker 2000). Occupational exposure to chromium occurs mainly from chromate production, stainless steel production and welding,

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Table 6-5. Chromium Content in Tissues and Body Fluids of the General Population

Sample	Median or mean	Range	Reference
Serum	0.006 µg/L	0.01–0.17 µg/L	Sunderman et al. 1989
Urine	0.4 µg/L	0.24–1.8 µg/L	Iyengar and Woittiez 1988
Lung	201 µg/kg (wet weight)	28–898 µg/kg (wet weight)	Raithel et al. 1987
Breast milk	0.30 µg/L	0.06–1.56 µg/L	Casey and Hambidge 1984
Hair	0.234 mg/kg	Not available	Takagi et al. 1986
Nail	0.52 mg/kg	No applicable	Takagi et al. 1988

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chromium plating, ferrochrome alloys and chrome pigment production, and working in tanning industries (Ashley et al. 2003). A list of industries that may be sources of chromium exposure is given in Table 6-6. For most occupations, exposure is due to both chromium(III) and chromium(VI) present as soluble and insoluble fractions. However, exceptions include the tanning industry, where exposure is mostly from soluble chromium(III), and the plating industry, where exposure is due to soluble chromium(VI). The typical concentration ranges of airborne chromium(VI) to which workers in these industries were exposed during an average of 5–20 years of employment were: chromate production, 100–500 $\mu\text{g}/\text{m}^3$; stainless steel welding, 50–400 $\mu\text{g}/\text{m}^3$; chromium plating, 5–25 $\mu\text{g}/\text{m}^3$; ferrochrome alloys, 10–140 $\mu\text{g}/\text{m}^3$; and chrome pigment, 60–600 $\mu\text{g}/\text{m}^3$ (Stern 1982). In the tanning industry, except for two bath processes, the typical exposure range due to chromium(III) was 10–50 $\mu\text{g}/\text{m}^3$. A study of chromium(VI) levels of in the air of a chrome plating shop measured concentrations of chromium(VI) in the range of 10–30 $\mu\text{g}/\text{m}^3$ for chrome plating shops with local exhaust (Pellerin and Booker 2000). In plating shops without local exhaust, the levels were much higher, up to 120 $\mu\text{g}/\text{m}^3$ (Pellerin and Booker 2000). In an occupational exposure study of chromium in an aircraft construction factory, airborne samples were collected over a 4-hour period; urinary samples were collected at the beginning (Monday), end (Friday), and after the work shift in order to analyze the absorption of chromium during working hours (Gianello et al. 1998). The air sampling results were 0.02–1.5 mg/m^3 , and the urine sampling results were 0.16–7.74 $\mu\text{g}/\text{g}$ creatinine. Compared to the ACGIH and BEI-ACGIH Hygiene Standard of 50 $\mu\text{g}/\text{m}^3$, both sets of results indicated a very low risk of exposure. The National Occupational Exposure Survey (NOES) conducted by NIOSH from 1981 to 1983 estimated that 304,829 workers in the United States were potentially exposed to chromium(VI) (NIOSH 1989). The NOES database does not contain information on the frequency, concentration, or duration of exposure; the survey only estimates the number of workers potentially exposed to chemicals in the workplace.

In a survey of workers in pigment factories in England that produced strontium and lead chromate, the concentrations of chromium in the whole blood in exposed workers ranged from 3 to 216 $\mu\text{g}/\text{L}$, compared to a level of <1 $\mu\text{g}/\text{L}$ for the nonoccupationally exposed population (McAughey et al. 1988). The corresponding concentrations in the urine of exposed workers and the unexposed population were 1.8–575 μg chromium/g creatinine and <0.5 μg chromium/g creatinine, respectively (McAughey et al. 1988).

Other investigators have found a higher lung burden for chromium in occupational groups than in unexposed groups. The median concentration of chromium in the lungs of deceased smelter workers in Sweden was 450 $\mu\text{g}/\text{kg}$ (wet weight), compared to a value of 110 $\mu\text{g}/\text{kg}$ (wet weight) for rural controls and 199 $\mu\text{g}/\text{kg}$ (wet weight) for urban controls (Gerhardsson et al. 1988).

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Table 6-6. Industries that May be Sources of Chromium Exposure

Abrasives manufacturers	Laboratory workers
Acetylene purifiers	Leather finishers
Adhesives workers	Linoleum workers
Aircraft sprayers	Lithographers
Alizarin manufacturers	Magnesium treaters
Alloy manufactures	Match manufacturers
Aluminum anodizers	Metal cleaners
Anodizers	Metal workers
Battery manufacturers	Milk preservers
Biologists	Oil drillers
Blueprint manufacturers	Oil purifiers
Boiler scalers	Painters
Candle manufacturers	Palm-oil bleachers
Cement workers	Paper water proofers
Ceramic workers	Pencil manufacturers
Chemical workers	Perfume manufacturers
Chromate workers	Photoengravers
Chromium-alloy workers	Photographers
Chromium-alum workers	Platinum polishers
Chromium platers	Porcelain decorators
Copper etchers	Pottery frosters
Copper-plate strippers	Pottery glazers
Corrosion-inhibitor workers	Printers
Crayon manufacturers	Railroad engineers
Diesel locomotive repairmen	Refractory-brick manufacturers
Drug manufacturers	Rubber manufacturers
Dye manufacturers	Shingle manufacturers
Dyers	Silk-screen manufacturers
Electroplaters	Smokeless-powder manufacturers
Enamel workers	Soap manufacturers
Explosive manufacturers	Sponge bleachers
Fat purifiers	Steel workers
Fireworks manufacturers	Tanners
Flypaper manufacturers	Textile workers
Furniture polishers	Wallpaper printers
Fur processors	Wax workers
Glass-fibre manufacturers	Welders
Glue manufacturers	Wood-preservative workers
Histology technicians	Wood stainers
Jewelers	

Source: IARC 1990

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A study of seasonal workers who manually replant trees in harvested forests in British Columbia, Canada, examined whether there were potentially higher exposures to arsenic, lead, cadmium, chromium, and nickel for workers who used fertilizer relative to tree planters who did not use fertilizer (Gorman et al. 2011). The study concluded that there was no evidence to suggest that tree planters who worked with fertilizer were at an elevated risk of exposure to these metals as compared to those who did not use fertilizer.

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.

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 ingest inappropriate materials (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).

Children living in vicinities where there are chromium waste sites nearby may be exposed to chromium to a greater extent than adults through inhalation of chromium particulates and through contact with contaminated soils. One study has shown that the average concentration of chromium in the urine of children at ages five and younger was significantly higher than in adults residing near sites where chromium waste slag was used as fill material (Fagliano et al. 1997), and the soil levels of a hazardous waste disposal site in New Jersey were measured at levels up to $120 \mu\text{g}/\text{m}^3$ (Pellerin and Booker 2000). The tendency of young children to ingest soil, either intentionally through pica or unintentionally through hand-to-mouth activity, is well documented and can result in ingestion of chromium present in soil and dust. Soil may affect the bioavailability of contaminants in several ways, most likely by acting as a competitive sink for the contaminants. In the presence of soil, the contaminants will partition between absorption by the gut and sorption onto the soil particles (Sheppard et al. 1995). If the contaminant is

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irreversibly bound to soil particles, then the contaminant is unlikely to be absorbed in the gastrointestinal tract. Hexavalent chromium exists in soils as a relatively soluble anion and may be present in bioavailable form, possibly with enhanced absorption due to the presence of the soil itself. In contrast, chromium(III) present in soil is generally not very soluble or mobile under most environmental conditions and is not readily bioavailable (James et al. 1997). Studies discussing the oral absorption of chromium in rats from a soil surface in which 30% of the chromium was in hexavalent form and 70% was in trivalent form suggested that while absorption in animals is quite low, chromium appeared to be better absorbed from soil than from soluble chromate salts (Witmer et al. 1989, 1991). However, less than half of the administered dose of chromium could be accounted for in this study, and in separate experiments with low dosages administered to the rats, the control animals actually had higher concentrations of chromium than the animals that were administered the oral dose. Children may accidentally ingest chromium picolinate in households whose members use this product as a dietary supplement unless it is well stored and kept away from children. Small amounts of chromium are used in certain consumer products such as toners in copying machines and printers, but childhood exposure from these sources is expected to be low. Children may also be exposed to chromium from parents' clothing or items removed from the workplace if the parents are employed in a setting where occupational exposure is significant (see Section 6.5). Chromium has been detected in breast milk at concentrations of 0.06–1.56 µg/L (Casey and Hambidge 1984), suggesting that children could be exposed to chromium from breast-feeding mothers. Studies on mice have shown that chromium crosses the placenta and can concentrate in fetal tissue (Danielsson et al. 1982; Saxena et al. 1990a).

A study done on the potential exposure of teenagers to airborne chromium from steel dust in the New York City subway system found significantly higher concentrations of chromium than in home and ambient samples. The conclusion from the study was that the increased concentration was most likely due to steel dust present in the subway system as the source of chromium (Chillrud et al. 2004). Chromium levels in the New York City subway system are greater than ambient levels by approximately two orders of magnitude. Chromium levels observed in the study (≈ 84 ng/m³) are similar to chronic reference guidelines in both Canada and the United States and were 40–100 times the adult range in the estimated 10^{-5} lifetime cancer risk (Chillrud et al. 2004). The reference values for exposure to chromium range from 2 to 100 ng/m³ (Wu et al. 2001). The study measured total chromium levels, without separating the species of chromium into chromium(III) and chromium(VI). Previous studies have suggested that airborne chromium generated from steel welding have a significant amount of chromium(VI) present, extending the possibility that there is a possibility for chromium(VI) to be present in the steel dust in the New York City subway system as well (Chillrud et al. 2004; Edme et al. 1997).

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Another study done on the bones of deceased neonatal humans in Poland found that statistically significant differences in chromium concentrations were observed. In addition, a positive correlation between pairs of metals was observed, specifically between the pairing of chromium and lead. (Baranowski et al. 2002). Bones were chosen to examine, since they are a useful reference in regards to heavy metal exposure and accumulation, and are therefore an accurate measure of chronic exposure.

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

In addition to individuals who are occupationally exposed to chromium (see Section 6.5), there are several groups within the general population that have potentially high exposures (higher-than-background levels) to chromium. Persons using chromium picolinate as a dietary supplement will also be exposed to higher levels of chromium than those not ingesting this product (Anderson 1998b). Like many other products used to promote weight loss or speed metabolism, there is also the potential for overuse of this product by some members of the population in order to achieve more dramatic results (Wasser et al. 1997). People may also be exposed to higher levels of chromium if they use tobacco products, since tobacco contains chromium (IARC 1980).

Workers in industries that use chromium are one segment of the population that is especially at high risk to chromium exposure. Many industrial workers are exposed to chromium(VI) levels in air that exceed the accepted occupational exposure limits (Blade et al. 2007). Occupational exposure from chromate production, stainless steel welding, chromium plating, and ferrochrome and chrome pigment production is especially significant since the exposure from these industries is to chromium(VI). Occupational exposure to chromium(III) compounds may not be as great a concern as exposure to chromium(VI) compounds. Among the general population, residents living near chromate production sites may be exposed to higher levels of chromium(VI) in air. Ambient concentrations as high as $2.5 \mu\text{g}/\text{m}^3$ chromium in air were detected in a 1977 sample from Baltimore, Maryland (EPA 1984a). People who live near chromium waste disposal sites and chromium manufacturing and processing plants may be exposed to elevated levels of chromium. The airborne concentrations of chromium(VI) and total chromium in a contaminated site in Hudson County, New Jersey were studied (Falerios et al. 1992). The mean concentrations of both chromium(VI) and total chromium in indoor air of the contaminated site were about three times higher than the mean indoor air concentrations of uncontaminated residential sites in Hudson County. Although the mean concentration of chromium(VI) in outdoor air was much lower than the current occupational exposure limit of $50 \mu\text{g}/\text{m}^3$, levels in 10 of 21 samples at the contaminated site

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exceeded the background urban outdoor chromium(VI) concentration of 4 ng/m^3 . Similarly, the total chromium concentration in 11 of 21 outdoor air samples from the contaminated site exceeded the outdoor mean concentration of 15 ng/m^3 for urban New Jersey. However, recent sampling data from Hudson County, New Jersey have shown that more than two-thirds of previously sampled sites contaminated with chromite ore processing residue did not have statistically significant mean concentrations greater than the background levels (Scott et al. 1997a). These data, as well as the results of a soil dispersion model (Scott et al. 1997b), suggest that heavy vehicular traffic over unpaved soil surfaces containing chromium(VI) are required for high levels of atmospheric chromium(VI) at these sites. Persons using contaminated water for showering and bathing activities may also be exposed via inhalation to potentially high levels of chromium(VI) in airborne aerosols (Finley et al. 1996a). In a field study to simulate daily bathing activity, airborne chromium(VI) concentrations were about 2 orders of magnitude greater than ambient outdoor air concentrations when water concentrations of 5.4 and 11.5 mg/L were used in the shower.

A study was conducted from September to November 1989 to determine the levels of chromium in urine and red blood cells of state employees who worked at a park (with only indirect exposure potential) adjacent to chromium-contaminated sites in Hudson County, New Jersey (Bukowski et al. 1991). The chromium levels in red blood cells and urine of 17 of these employees showed no differences compared to 36 employees who worked at state parks outside Hudson County. The authors concluded that urinary and blood levels of chromium are poor biological markers in gauging low-level environmental exposure to chromium. This study also concluded that chromium levels in blood and urine depended on other confounding variables, such as exercise, past employment in a chromium-exposed occupation, beer drinking, and diabetic status. Other lifestyle (e.g., smoking), dietary, or demographic factors had no measurable effect on blood and urinary chromium. These conclusions are consistent with the results of a study that measured the urinary excretion of chromium following oral ingestion of chromite ore processing residue material for three days (Finley and Paustenbach 1997). These results indicate no statistical difference in mean urinary chromium concentrations in groups of individuals exposed to chromite ore processing residue material versus the control group. High levels of chromium were detected in the urine and hair of individuals living near a chromite ore-processing plant in Mexico (Rosas et al. 1989), which suggests the possibility of using these media as biological markers in gauging long term, high-level environmental exposure to chromium.

Elevated levels of chromium in blood, serum, urine, and other tissues and organs have been observed in patients with cobalt-chromium knee and hip arthroplasts (Coleman et al. 1973; Michel et al. 1987; Sunderman et al. 1989).

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The chromium content in cigarette tobacco from the United States has been reported to be 0.24–6.3 mg/kg (IARC 1980), but neither the chemical form nor the amount of chromium in tobacco smoke is known. People who use tobacco products may be exposed to higher-than-normal levels of chromium.

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 chromium 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 chromium.

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. As seen in Section 4.2, the relevant physical and chemical properties of chromium and its compounds are known (Hartford 1979; NIOSH 2008; Weast 1985) and prediction of environmental fate and transport of chromium in environmental media is possible. However, the physical or chemical forms and the mode by which chromium(III) compounds are incorporated into biological systems are not well characterized. The determination of the solubilities of hexavalent chromium compounds in relevant body fluids (e.g., the solubility of chromates in lung fluid) may also be helpful.

Production, Import/Export, Use, Release, and Disposal. Knowledge of a chemical's production volume is important because it may indicate environmental contamination and human exposure. If a chemical's production volume is high, there is an increased probability of general population exposure via consumer products and environmental sources, such as air, drinking water, and food. Data concerning the production (Hartford 1979; Papp and Lipin 2001; SRI 1997; USGS 2008b),

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import (USGS 2008b), and use (CMR 1988a, 1988b; EPA 1984a; IARC 1990; Papp and Lipin 2001; USDI 1988a; USGS 2008b) of commercially significant chromium compounds are available. Chromium is not generally used to process foods for human consumption or added to foods other than diet supplements. Thus, consumer exposure to chromium occurs mostly from natural food sources (Bennett 1986; EPA 1984a; Kumpulainen et al. 1979), but this exposure will increase particularly for people who consume acidic food cooked in stainless steel utensils (Anderson 1981; EPA 1984a). Exposure to chromium occurs to a much lesser extent from products such as toners of photocopying machines, some wood treatment chemicals, and through other chromium-containing consumer products (CMR 1988a, 1988b; EPA 1984a; IARC 1990; USDI 1988a).

As seen in Tables 6-1 and 6-2, the largest amount of chromium from production and user facilities is disposed of on land or transferred to an off-site location. More detailed site-and medium-specific (e.g., air, water, or soil) release data for chromium that is disposed of off-site are necessary to assess the exposure potential to these compounds from different environmental media and sources. There are EPA guidelines regarding the disposal of chromium wastes and OSHA regulations regarding the levels of chromium in workplaces (EPA 1988a; OSHA 1998a).

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 2009, became available in May of 2011. This database is updated yearly and should provide a list of industrial production facilities and emissions.

Environmental Fate. Information is available to permit assessment of the environmental fate and transport of chromium in air (Pacyna and Pacyna 2001; Schroeder et al. 1987; Scott et al. 1997a, 1997b), water (Cary 1982; Comber and Gardner 2003; EPA 1980, 1984a; Fishbein 1981; Schmidt and Andren 1984; WHO 2003) and soil (Ashley et al. 2003; Avudainayagam et al. 2003; BalasoIU et al. 2001; Bartlett 1991; Calder 1988; Cary 1982; Jardine et al. 1999; Rifkin et al. 2004). Chromium is primarily removed from the atmosphere by fallout and precipitation. The residence time of chromium in the atmosphere is expected to be <10 days (Nriagu 1979). Most of the chromium in lakes and rivers will ultimately be deposited in the sediments. Chromium in the aquatic phase occurs in the soluble state or as suspended solids adsorbed onto clayish materials, organics, or iron oxides (Cary 1982). Most of the soluble chromium is present as chromium(VI) or as soluble chromium(III) complexes and generally accounts for a small percentage of the total (Cary 1982). Additional data, particularly regarding chromium's nature of speciation, would be useful to fully assess chromium's fate in air. For example, if chromium(III) oxide

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forms some soluble salt in the air due to speciation, its removal by wet deposition will be faster. No data regarding the half-life of chromium in the atmosphere or a measure of its persistence are available. In aquatic media, sediment will be the ultimate sink for chromium, although soluble chromates may persist in water for years (Cary 1982; EPA 1984a). Additional data elucidating the nature of speciation of chromium in water and soil would also be desirable and a direct measurement of the chromium residence time in the atmosphere would be useful.

Bioavailability from Environmental Media. The bioavailability of chromium compounds from contaminated air, water, soil, or plant material in the environment has not been adequately studied. Absorption studies of chromium in humans and animals provide information regarding the extent and rate of inhalation (Cavalleri and Minoia 1985; Kiilunen et al. 1983; Langård et al. 1978) and oral exposure (Anderson 1981, 1986; Anderson et al. 1983; Donaldson and Barreras 1966; Randall and Gibson 1987; Suzuki et al. 1984). A sorption study has measured the amount of chromium(VI) when iron particles are present in the water samples; the conclusion was that soluble chromium(VI) present in the water could sorb on to iron particles present in the acidic environment of the stomach, and thus, be less bioavailable (Parks et al. 2004). These available absorption studies indicate that chromium(VI) compounds are generally more readily absorbed from all routes of exposure than are chromium(III) compounds. This is consistent, in part, with the water solubilities of these compounds (Bragt and van Dura 1983). The bioavailability of both forms is greater from inhalation exposure than from ingestion or dermal exposure. The bioavailability of chromium from soil depends upon several factors (Witmer et al. 1989). Factors that may increase the mobility of chromium in soils include the speculated conversion of chromium(III) to chromium(VI), increases in pH, and the complexation of chromium(III) with organic matter from water-soluble complexes. Data on the bioavailability of chromium compounds from actual environmental media and the difference in bioavailability for different media need further development.

Food Chain Bioaccumulation. It is generally believed that chromium does not bioconcentrate in fish (EPA 1980, 1984a; Fishbein 1981; Schmidt and Andren 1984) and there is no indication of biomagnification of chromium along the aquatic food chain (Cary 1982). However, recent skin biopsy data indicate that North Atlantic right whales are exposed to hexavalent chromium and accumulate a range of 4.9–10 µg chromium/g tissue with a mean of 7.1 µg chromium/g tissue (Wise et al. 2008). Some data indicate that chromium has a low mobility for translocation from roots to aboveground parts of plants (Cary 1982; WHO 1988). However, more data regarding the transfer ratio of chromium from soil to plants and biomagnification in terrestrial food chains would be desirable.

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Exposure Levels in Environmental Media. The atmospheric total chromium concentration in the United States is typically $<10 \text{ ng/m}^3$ in rural areas and $10\text{--}30 \text{ ng/m}^3$ in urban areas (Fishbein 1984; WHO 2003). Most drinking water supplies in the United States contain $<5 \text{ }\mu\text{g/L}$ of chromium (WHO 2003). The chromium level in soils varies greatly and depends on the composition of the parent rock from which the soils were formed. Basalt and serpentine soils, ultramafic rocks, and phosphorites may contain chromium as high as a few thousand mg/kg (Merian 1984), whereas soils derived from granite or sandstone will have lower concentrations of chromium (Swaine and Mitchell 1960). The concentration range of chromium in 1,319 samples of soils and other surficial materials collected in the conterminous United States was $1\text{--}2,000 \text{ mg/kg}$, with a geometric mean of 37 mg/kg (USGS 1984). There is a large variation in the available data regarding the levels of chromium in foods (EPA 1984a). Concentrations ranges are $30\text{--}230 \text{ }\mu\text{g/kg}$ in vegetables, $20\text{--}510 \text{ }\mu\text{g/kg}$ in fruits, $40\text{--}220 \text{ }\mu\text{g/kg}$ in grains and cereals, and $110\text{--}230 \text{ }\mu\text{g/kg}$ in meats and fish (EPA 1984a). It would be useful to develop nationwide monitoring data on the levels of chromium in U.S. ambient air and drinking water, and these data should quantitate levels of both chromium(III) and chromium(VI) and not just total chromium. EPA is working with state and local officials to determine the prevalence of chromium (VI) in drinking water (Erickson 2011; Kemsley 2011).

Exposure Levels in Humans. The general population is exposed to chromium by inhaling ambient air and ingesting food and drinking water containing chromium. Dermal exposure of the general public to chromium can occur from skin contact with certain consumer products that contain chromium or from contact with chromium contaminated soils. Some of the consumer products known to contain chromium are certain wood preservatives, cement, cleaning materials, textiles, and leather tanned with chromium (WHO 1988). However, no quantitative data for dermal exposure to chromium-containing consumer products were located. Levels of chromium in ambient air ($<0.01\text{--}0.03 \text{ }\mu\text{g/m}^3$) (Fishbein 1984; WHO 2003) and tap water ($<2 \text{ }\mu\text{g/L}$) (WHO 2003) have been used to estimate the daily intake of chromium via inhalation ($<0.2\text{--}0.6 \text{ }\mu\text{g}$) and tap water ($<4 \text{ }\mu\text{g}$). These estimates are based on an air inhalation rate of $20 \text{ m}^3/\text{day}$ and a drinking water consumption rate of 2 L/day . The daily chromium intake for the U.S. population from consumption of selected diets (diets with 25 and 43% fat) has been estimated to range from 25 to $224 \text{ }\mu\text{g}$, with an average of $76 \text{ }\mu\text{g}$ (Kumpulainen et al. 1979). This value is within the range established by the World Health Organization (WHO) as a mean chromium intake from food and water of $52\text{--}943 \text{ }\mu\text{g/day}$ (WHO 2003). However, few data on the levels of chromium in body tissues or fluids for populations living near hazardous waste sites are available. Such data could be a useful tool as an early warning system against harmful exposures. In addition, there is a need for data on the background levels

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of chromium in body fluids of children. Such data would be important in assessing the exposure levels of this group of people.

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

Exposures of Children. Limited data exist regarding exposure and body burdens of chromium in children. Chromium has been detected in breast milk at concentrations of 0.06–1.56 µg/L (Casey and Hambidge 1984), suggesting that children could be exposed to chromium from breast-feeding mothers. Studies in mice have shown that chromium crosses the placenta and can concentrate in fetal tissue (Danielsson et al. 1982; Saxena et al. 1990a). Because children living near areas contaminated with chromium have been shown to have elevated chromium levels in urine as compared to adults (Fagliano et al. 1997), additional body burden studies are required to evaluate the exposures and the potential consequences that this might have upon children. This is particularly important around heavily contaminated soils where children may be exposed dermally or through inhalation of soil particulates during play activities. These studies may determine if children may be more susceptible than adults to the toxic effects of chromium including immunosensitivity. Studies are necessary that examine children's weight-adjusted intake of chromium and determine how it compares to that of adults. Since chromium is often detected in soil surfaces and children ingest soil either intentionally through pica or unintentionally through hand-to-mouth activity, pica is a unique exposure pathway for children. Studies have shown that although absorption of chromium is low, it may be enhanced slightly from contaminated soil surfaces (Witmer et al. 1989, 1991).

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

Exposure Registries. No exposure registries for chromium 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.

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6.8.2 Ongoing Studies

The Federal Research in Progress (FEDRIP 2008) 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 (see Table 6-7).

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Table 6-7. Ongoing Studies on Chromium

Principal investigator	Affiliation	Research description
David Brautigam	University of Virginia	Define the biochemical basis of chromium enhancement of insulin action
William Cefalu	Pennington Biomedical Research Center	Study the changes in insulin uptake on chromium supplementation
Mitchell Cohen	New York University School of Medicine	Study biological interactions of metals and improve design of metallopharmaceuticals
Jeffrey Elmendorf	Indiana University: Purdue University at Indianapolis	Chromium action and role in the glucose transport system
Emily Horvath	Indiana University: Purdue University at Indianapolis	Cellular insulin resistance mechanisms
Joshua Jacobs	Rush University Medical Center	Metal release and effects in people with metal-on-hip replacements
Sushil Jain	Louisiana State University	Cytokine production; role of chromium in preventing oxidative stress
Umesh Masharani	University of California, San Francisco	The effects of chromium on insulin action
Mahmood Mozaffari	Medical College of Georgia	Effect of chromium on glucose metabolism
Charles Myers	Medical College of Wisconsin	Study the mechanisms of chromium(VI) in the human lung system
Patricia Opresko	University of Pittsburgh at Pittsburgh	Understand the mechanisms of genomic instability associated with aging
Viresh Rawal	University of Chicago	Investigation of metal-salen complexes for use in C-C bond forming reactions; Diels-Alder catalyst development
James Rigby	Wayne State University	Study metal mediated cyclo-addition reactions to synthesize natural products
Diane Stearns	Northern Arizona University	Study the difference in mutations caused by soluble chromium vs. insoluble chromium; discover mechanism of cellular entry by soluble chromium compounds
Kent Sugden	University of Montana	Study the role of chromium in DNA mutations and cancer
Bo Xu	Southern Research Institute	The effect of chromium exposure on DNA damage
Ziling Xue	University of Tennessee, Knoxville	Development of an analytical method to determine chromium levels in biological fluids
Anatoly Zhitkovich	Brown University	Study the role of chromium in genetic alteration of cells after exposure

Source: FEDRIP 2008