6. POTENTIAL FOR HUMAN EXPOSURE

6.1   OVERVIEW

Cesium has been identified in at least 8 of the 1,636 hazardous waste sites that have been proposed for inclusion on the EPA National Priorities List (NPL) (HazDat 2003). It was reported that ^{134}Cs has been found in at least 3 of the 1,636 current or former NPL sites and ^{137}Cs has been detected in at least 23 of the 1,636 current or former NPL sites. However, the number of sites evaluated for cesium is not known. The frequency of these sites can be seen in Figures 6-1, 6-2, and 6-3. Of these sites, none are located in the Commonwealth of Puerto Rico.

Naturally-occurring cesium and cesium minerals consist of only one stable isotope, ^{133}Cs. Cesium occurs in the earth's crust at low concentrations. Granites contain an average cesium concentration of about 1 ppm and sedimentary rocks contain about 4 ppm (Burt 1993). Higher concentrations are found in lepidolite, carnallite, muscovite, beryl, spodumene, potassium feldspars, leucite, petalite, and related minerals. The most important source of commercial cesium is the mineral pollucite, which usually contains about 5–32% Cs_2O (Burt 1993). The largest deposits of pollucite are located in Manitoba, Canada and account for about two-thirds of the world's known supply. Smaller deposits are located in Zimbabwe, Namibia, Brazil, Scandinavia, Czechoslovakia, and the United States. Continental dust and soil erosion are the main emission sources of naturally occurring cesium present in the environment. Cesium is also released to the environment as a result of human activities. The mining of pollucite ores and the production and use of cesium compounds in the electronic and energy production industries contribute to its direct release to the environment. Cesium has also been detected in the fly ash of hazardous waste incinerators and coal burning power plants (Fernandez et al. 1992; Mumma et al. 1990). Since the production and use of cesium compounds are limited, and since the natural concentration of cesium in the earth's crust is low, stable ^{133}Cs is not often monitored or detected in the environment.

Of much greater concern is the release of radioactive forms of cesium to the environment, such as ^{137}Cs and ^{134}Cs. These and other radioactive isotopes were released to the environment as a result of atmospheric testing of nuclear weapons (carried out from 1945 to 1980) and accidents that occurred at nuclear power plants such as the incident at the Chernobyl nuclear power plant in 1986 and the accident at the Windscale nuclear weapons facility in the United Kingdom in 1957. Small amounts of ^{137}Cs and ^{134}Cs
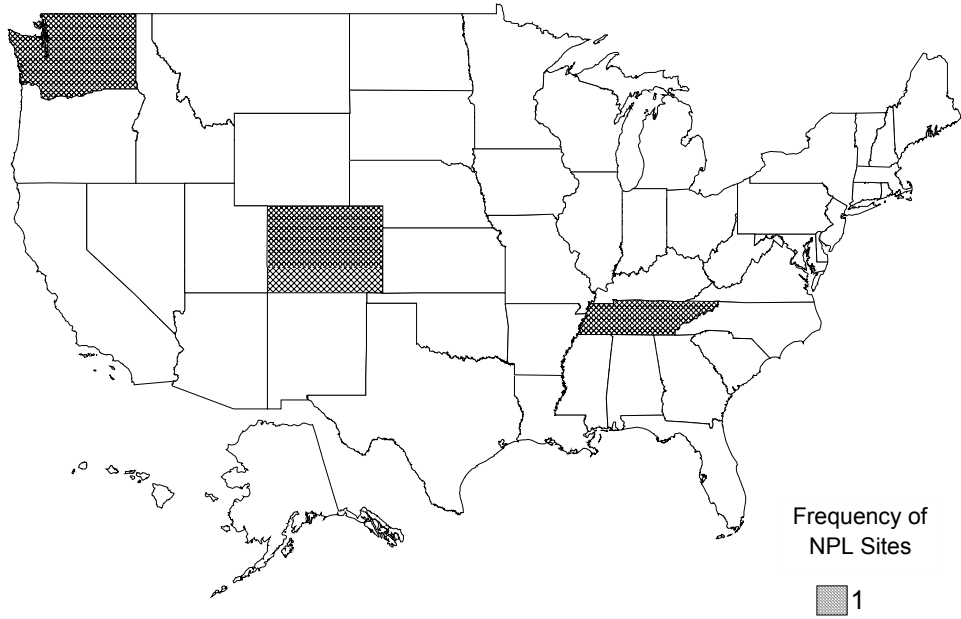
Figure 6-1. Frequency of NPL Sites with Cesium Contamination

Frequency of
NPL Sites

1
2

Derived from HazDat 2003

Figure 6-2. Frequency of NPL Sites with ^{134}Cs Contamination

Frequency of
NPL Sites

1

Derived from HazDat 2003

6. POTENTIAL FOR HUMAN EXPOSURE

Figure 6-3. Frequency of NPL Sites with ^{137}Cs Contamination

Frequency of
NPL Sites

	1
	2
	3
	5

Derived from HazDat 2003

are also released in the airborne and liquid effluents during the normal operation of nuclear power plants. These levels are very low in comparison to the amounts released from weapons tests and accidents at nuclear power plants and are not expected to have a major impact upon human health. For the most part, testing of nuclear weapons has been discontinued by most nations for many years now. However, India and Pakistan have recently (May of 1998) conducted limited underground tests of nuclear weapons (UN 1998).

Radioactive materials are commonly referred to by activity rather than mass. The activity is the number of transformations (disintegrations) that the material undergoes in a given period of time. The most common units of activity are the curie (Ci) or the becquerel (Bq). One curie is equal to 3.7×10^{10} transformations per second (tps) and 1 becquerel is equal to 1 tps. For convenience, picocuries (pCi) are often reported for lower activities; $1 pCi = 1 \times 10^{-12} Ci = 0.037 Bq$.

Radioactive cesium is removed from the air by wet and dry deposition and can travel thousands of miles before settling to earth. Wet deposition is considered the most important pathway for the removal of radioactive cesium from the atmosphere. It is a complex process that depends upon meteorological conditions such as temperature, the microphysical structure of the clouds, and the rainfall rate, as well as the physical and chemical properties of the airborne cesium.

Cesium has very low mobility in soil. In general, it has been reported that cesium usually does not migrate below depths of about 40 cm, with the majority of cesium being retained in the upper 20 cm of the soil surface (Korobova et al. 1998; Takenaka et al. 1998). Clay minerals and soils rich in exchangeable potassium adsorb cesium by binding the cations to interlayer positions of the clay particles (Paasikallio 1999). The low hydration energy of cesium cations is primarily responsible for their selective sorption and fixation by clays. These factors can limit the uptake of cesium in grass and plant material. There are exceptional areas, however, where cesium fixation in soil is much less, resulting in greater transport in the soil and uptake in plants. Regions in Venezuela, Brazil, and Russia have been identified where the mobility of cesium is considerably greater than in other soils (LaBrecque and Rosales 1996; WHO 1983). Cesium is also deposited on plants and trees by wet and dry deposition and is absorbed into the flora through its foliage (Sawidis et al. 1990). The deposited cesium can make its way to soil through decomposition of the contaminated foliage.

Since the half-life for some radioactive isotopes of cesium is long (the half-life of ^{137}Cs is about 30 years and the half-life of [134]Cs is about 2 years), the general population is exposed to ^{137}Cs and [134]Cs for long

periods of time after it is released from a nuclear accident or weapons test, with the greatest exposure occurring near the source. Although inhalation and dermal exposure is possible, oral ingestion of contaminated food items is the greatest source of internal exposure for both naturally occurring and radioactive cesium. Cesium is uniformly distributed throughout the whole body similar to potassium, and it does not accumulate in any one particular part of the body like iodine (thyroid) or strontium (bones). For this reason, radioactive cesium only poses significant health risks if a large amount has been ingested. Workers employed in the mining and milling of pollucite ores and the production of cesium compounds are exposed to cesium through oral, dermal, and inhalation routes. Similar routes of exposure to ^{137}Cs and ^{134}Cs occurs for workers employed in the nuclear industry. External exposure to gamma radiation can also occur for workers employed in the nuclear industry as well as for the general population. The health consequences of external exposure to gamma radiation are not unique to ^{137}Cs and ^{134}Cs, but are similar for all gamma emitting radionuclides.

6.2   RELEASES TO THE ENVIRONMENT

Throughout this chapter, the units used to express concentration or intakes of cesium are the same units reported by the authors. In most cases, values are expressed in mass units when referring to ^{133}Cs, while radioactive cesium isotopes are expressed in units of activity.

According to the Toxics Release Inventory (TRI), in 2000, there were no reportable releases of ^{133}Cs, or its compounds into the environment by commercial sources (TRI00 2002). The TRI data should be used with caution since only certain types of facilities are required to report. Therefore, it is not an exhaustive list. Facilities are required to report to TRI if they have 10 or more full-time employees, or if the facility is classified under Standard Industrial Classification (SIC) codes 20–39, if the facility manufactures or processes over 25,000 pounds of the chemical, or otherwise uses >10,000 pounds of the chemical in a calendar year.

In the United States, commercial nuclear power plant operators are required to report any detectable quantities of radioactive materials released to the environment (10 CFR 50.36a). Table 6-1 summarizes releases of ^{137}Cs and ^{134}Cs to the atmosphere and water for 1993 from pressurized water reactor (PWR) and boiling water reactor (BWR) nuclear power plants. Nearly all of the radioactive material reported as being released in effluents are from planned releases. Planned releases result from normal plant operation or from anticipated operational occurrences. The latter include unplanned releases of radioactive

Table 6-1. Radiocesium Releases from Nuclear Power Plants for 1993

| | | Annual total site environmental releases for 1993 | | | |
| | | Water | | Air | |
Installation	Location[a]	^{134}Cs, Ci	^{137}Cs, Ci	^{134}Cs, Ci	^{137}Cs, Ci
Boiling water reactors					
Browns Ferry[b]	Decatur, AL	0.033	0.18	3.5×10^{-4}	8.9×10^{-4}
Brunswick[b]	Wilmington, NC	2.9×10^{-4}	1.7×10^{-3}	No data	7.0×10^{-4}
Clinton	Clinton, IL	No data	No data	No data	No data
Cooper	Omaha, NE	9.3×10^{-3}	0.052	No data	No data
Dresden[b]	Joliet, IL	1.2×10^{-6}	0.025	No data	1.9×10^{-4}
Duane Arnold	Cedar Rapids, IA	No data	No data	No data	1.4×10^{-6}
Edwin I. Hatch	Baxley, GA	6.3×10^{-3}	0.044	No data	6.2×10^{-5}
Fermi	Laguna Beach, MI	No data	8.3×10^{-6}	No data	No data
Grand Gulf	Vicksburg, MS	3.5×10^{-4}	6.0×10^{-4}	2.1×10^{-6}	2.3×10^{-6}
Hope Creek	Wilmington, DE	No data	4.8×10^{-5}	No data	No data
Humbolt Bay[b]	Eureka, CA	No data	8.8×10^{-3}	No data	3.2×10^{-5}
James A. Fitzpatrick	Syracuse, NY	No data	4.1×10^{-5}	No data	6.6×10^{-6}
LaCrosse[b]	LaCrosse, WI	3.2×10^{-4}	0.010	No data	1.0×10^{-4}
LaSalle	Ottawa, IL	No data	No data	No data	No data
Limerick	Philadelphia, PA	2.0×10^{-3}	6.0×10^{-3}	No data	No data
Millstone	New London, CT	0.066	0.224	6.9×10^{-6}	8.9×10^{-5}
Monticello	St. Cloud, MN	No data	No data	No data	4.8×10^{-4}
Nine Mile Point	Oswego, NY	No data	No data	No data	1.9×10^{-5}
Oyster Creek	Toms River, NJ	No data	No data	No data	6.5×10^{-5}
Peach Bottom	Lancaster, PA	5.2×10^{-4}	1.4×10^{-3}	6.1×10^{-5}	3.0×10^{-4}
Perry	Painesville, OH	8.8×10^{-5}	1.6×10^{-4}	No data	No data
Pilgram	Boston, MA	No data	9.8×10^{-4}	2.4×10^{-5}	7.3×10^{-5}
Quad-Cites	Moline, IL	No data	3.6×10^{-3}	No data	1.8×10^{-4}
River Bend	Baton Rouge, LA	2.6×10^{-4}	2.2×10^{-3}	No data	No data
Shoreham	Brookhaven, NY	No data	No data	No data	No data
Susquehanna	Berwick, PA	No data	2.9×10^{-5}	No data	No data
Vermont Yankee	Brattleboro, VT	No data	No data	No data	9.9×10^{-5}
WNP-2	Richland, WA	6.3×10^{-3}	0.019	6.7×10^{-6}	7.2×10^{-6}
Total		**0.12**	**0.58**	**4.5×10^{-4}**	**3.3×10^{-3}**
Pressurized water reactors					
Arkansas One	Russellville, AR	0.054	0.11	No data	3.5×10^{-8}
Beaver Valley	Shippingport, PA	1.2×10^{-3}	2.3×10^{-3}	No data	No data
Big Rock Point	Charlevoix, MI	9.5×10^{-4}	0.012	1.1×10^{-5}	1.5×10^{-4}
Braidwood	Joliet, IL	6.5×10^{-4}	1.7×10^{-3}	No data	No data
Byron	Byron, IL	5.6×10^{-3}	1.5×10^{-3}	No data	No data
Callaway	Fulton, MO	5.3×10^{-4}	7.8×10^{-4}	No data	No data
Calvert Cliffs	Washington, DC	0.14	0.21	3.4×10^{-3}	4.2×10^{-3}

6. POTENTIAL FOR HUMAN EXPOSURE

Table 6-1. Radiocesium Releases from Nuclear Power Plants for 1993

Installation	Location[a]	Annual total site environmental releases for 1993			
		Water		Air	
		^{134}Cs, Ci	^{137}Cs, Ci	^{134}Cs, Ci	^{137}Cs, Ci
Catawba	Rock Hill, SC	1.6×10^{-3}	3.9×10^{-3}	No data	No data
Comanche Peak	Glen Rose, TX	8.7×10^{-3}	8.7×10^{-3}	No data	No data
Crystal River	Tampa, FL	2.7×10^{-3}	9.3×10^{-3}	No data	3.1×10^{-6}
Davis-Besse	Toledo, OH	1.4×10^{-3}	6.2×10^{-3}	No data	No data
Diablo Canyon	San Luis Obispo, CA	0.036	0.064	3.3×10^{-5}	2.4×10^{-4}
Donald C. Cook	St. Joseph, MI	1.3×10^{-3}	6.3×10^{-3}	No data	7.3×10^{-7}
Fort Calhoun	Omaha, NE	1.3×10^{-3}	0.012	No data	1.7×10^{-6}
H.B. Robinson	Hartsville, SC	3.6×10^{-4}	4.1×10^{-4}	5.8×10^{-6}	1.8×10^{-5}
Haddam Neck	Middletown, CT	0.013	0.023	8.0×10^{-4}	8.8×10^{-3}
Harris	Raleigh, NC	2.9×10^{-4}	3.9×10^{-4}	No data	No data
Indian Point[b]	Peekskill, NY	3.0×10^{-3}	0.10	No data	7.0×10^{-4}
Joseph M. Farley	Dothan, AL	1.5×10^{-3}	3.6×10^{-3}	No data	No data
Kewaunee	Green Bay, WI	No data	2.6×10^{-6}	No data	2.0×10^{-6}
Maine Yankee	Wicassett, ME	1.4×10^{-3}	8.2×10^{-3}	No data	4.2×10^{-5}
McGuire	Charlotte, NC	1.6×10^{-3}	4.6×10^{-3}	No data	1.3×10^{-6}
North Anna	NW Richmond, VA	6.2×10^{-3}	9.2×10^{-3}	3.4×10^{-6}	8.2×10^{-5}
Oconee	Greenville, SC	2.7×10^{-3}	0.010	No data	3.9×10^{-4}
Palisades	South Haven, MI	2.4×10^{-4}	4.3×10^{-3}	No data	1.1×10^{-5}
Palo Verde	Phoenix, AZ	No data	No data	2.0×10^{-3}	1.7×10^{-3}
Point Beach	Manitowoc, WI	0.019	0.027	6.9×10^{-3}	6.9×10^{-3}
Prairie Island	Minneapolis, MN	2.8×10^{-3}	3.8×10^{-3}	1.5×10^{-5}	1.8×10^{-5}
R.E. Ginna	Rochester, NY	0.042	0.041	No data	5.2×10^{-6}
Rancho Seco[b]	Sacramento, CA	1.9×10^{-5}	3.6×10^{-4}	No data	No data
Salem	Wilmington, DE	0.81	1.0	No data	7.0×10^{-7}
San Onofre[b]	San Clemente, CA	0.49	0.57	2.5×10^{-5}	4.0×10^{-5}
Seabrook	Portsmouth, NH	No data	3.3×10^{-5}	No data	No data
Sequoyah	Daisy, TN	0.086	0.14	No data	No data
South Texas	Bay City, TX	3.4×10^{-3}	5.4×10^{-3}	No data	No data
St. Lucie	Ft. Pierce, FL	0.055	0.083	9.2×10^{-6}	2.1×10^{-5}
Summer	Columbia, SC	1.6×10^{-3}	3.3×10^{-3}	2.2×10^{-5}	2.9×10^{-5}
Surry	Newport News, VA	7.6×10^{-5}	0.011	No data	7.2×10^{-5}
Three Mile Island[b]	Harrisburg, PA	0.026	0.030	1.2×10^{-7}	4.7×10^{-6}
Trojan[b]	Portland, OR	1.0×10^{-3}	4.0×10^{-3}	No data	No data
Turkey Point	Florida City, FL	9.4×10^{-4}	5.5×10^{-3}	No data	9.4×10^{-7}
Vogtle	Augusta, GA	5.6×10^{-3}	7.3×10^{-3}	No data	No data
Waterford	New Orleans, LA	0.013	0.016	No data	No data
Wolf Creek	Burlington, KS	0.022	0.024	No data	No data

Table 6-1. Radiocesium Releases from Nuclear Power Plants for 1993

| | | Annual total site environmental releases for 1993 | | | |
| | | Water | | Air | |
Installation	Location[a]	^{134}Cs, Ci	^{137}Cs, Ci	^{134}Cs, Ci	^{137}Cs, Ci
Yankee Rowe[b]	Greenfield, MA	4.3×10^{-6}	6.0×10^{-5}	No data	1.0×10^{-7}
Zion	Waukegan, IL	0.014	0.029	2.4×10^{-4}	2.9×10^{-4}
Total		**1.88**	**2.85**	**0.013**	**0.023**

[a]Post office state abbreviations used.
[b]Facilities that are permanently or indefinitely shut down.

materials from miscellaneous actions such as equipment failure, operator error, or procedure error; these releases are not of such consequence as to be considered an accident.

The TRI data should be used with caution because only certain types of facilities are required to report. This is not an exhaustive list.

6.2.1 Air

Stable cesium is introduced into the atmosphere by resuspension of soil, accidental release from mining and milling pollucite, and emissions from hazardous waste incinerators or coal burning plants. These emissions are expected to be low since cesium occurs naturally in the earth's crust at low concentrations and only small amounts of pollucite are mined annually. Cesium was detected at concentrations of 10.8 and 6.11 mg/m^3 in the effluent of a coal-burning power plant in the western United States (Ondov et al. 1989) and has been identified in the fly ash from municipal incinerators (Mumma et al. 1990, 1991). Fly ash from five municipal waste incinerators in the United States contained cesium at concentrations of 2,100–12,000 ppm (EPA 1990a). Stable cesium has been identified in air at one of the eight NPL hazardous waste sites where it was detected in some environmental media (HazDat 2003).

Radioactive isotopes of cesium such as ^{137}Cs and ^{134}Cs have been released to the atmosphere from atmospheric nuclear weapons testing, accidents at nuclear reactors, and nuclear-powered satellites burning up in the atmosphere upon re-entry. The total amount of ^{137}Cs released from weapons testing through 1980 was estimated as 2.6×10^7 Ci (9.6×10^{17} Bq), 76% of which was released in the northern hemisphere and 24% in the southern hemisphere (WHO 1983). On April 26, 1986, a steam buildup caused an explosion and fire at a nuclear power generating plant in Chernobyl, Russia, releasing an estimated 5.4×10^5 Ci (2.0×10^{16} Bq) of ^{134}Cs and 1.1×10^6 Ci (4.0×10^{16} Bq) of ^{137}Cs into the atmosphere over Europe (Watson 1987). Long-range atmospheric transport spread the radionuclides throughout the Northern Hemisphere. No airborne activity from Chernobyl has been reported south of the equator (Eisler 1995). By early May 1986, these radionuclides were readily detectable in environmental samples collected in North America (Huda et al. 1988). More recent estimates have put the total activity of ^{137}Cs released from the Chernobyl power plant as 2.3×10^6 Ci (8.5×10^{16} Bq) and 1.2×10^6 Ci (4.4×10^{16} Bq) for ^{134}Cs (Buzulukov and Dobrynin 1993). On January 24, 1978, the Soviet nuclear-powered satellite Cosmos 954 re-entered earth's atmosphere over the Canadian Arctic, releasing an estimated 86 Ci of ^{137}Cs (Barrie et al. 1992). In October 1957, an accident at the Windscale nuclear weapons plant at Sellafield in the United Kingdom resulted in the release of 595 Ci of ^{137}Cs (Agency for Toxic Substances and Disease

Registry 1999). Routine activities at nuclear power plants and fuel-reprocessing stations also release
^{137}Cs and ^{134}Cs to the environment on a regular basis. Radiocesium released in airborne effluents from
the normal operation of nuclear power plants is considered low in comparison to releases from past
atmospheric weapons tests and the major releases following accidents at nuclear power plants. In 1998, it
was reported that $1.3x10^{-4}$ Ci of ^{134}Cs and $5.1x10^{-3}$ Ci of ^{137}Cs were released to the atmosphere from the
Savannah River plutonium processing site in South Carolina (DOE 1998b). In 1993, the Nuclear
Regulatory Commission (NRC) estimated that 0.013 Ci of ^{134}Cs and 0.023 Ci of ^{137}Cs were released in
airborne effluents from 30 PWR nuclear power plants operating in the United States (USNRC 1993). It
was also estimated that $4.6x10^{-4}$ Ci of ^{134}Cs and $3.3x10^{-3}$ Ci of ^{137}Cs were released in airborne effluents
from 28 BWR nuclear power plants (USNRC 1993). The total airborne and liquid releases of ^{134}Cs and
^{137}Cs from the individual nuclear power plants are summarized in Table 6-1.

Radioactive ^{134}Cs was not identified in air at the 3 NPL hazardous waste sites where it was detected in
some environmental media, but ^{137}Cs was identified in air at 5 of the 23 NPL hazardous waste sites where
it was detected in some environmental media (HazDat 2003).

6.2.2   Water

Cesium can be released to water surfaces during the mining, milling, and production process of pollucite
ore. The natural erosion and weathering of rocks will also lead to cesium's introduction into ground and
surface water. Stable cesium was identified in groundwater and surface water at four and one of the eight
NPL hazardous waste sites, respectively, where it was detected in some environmental media (HazDat
2003).

The dumping of high and low level radioactive wastes into the Arctic waters by the former Soviet Union
has also led to the release of ^{137}Cs and ^{134}Cs as well as other radioactive nuclides into these waters. In the
past, the majority of radioactive cesium released to water surfaces in North America arose from
deposition following atmospheric nuclear weapons testing conducted by the United States, primarily
during the 1960s (Robbins et al. 1990). Radioactive cesium can be introduced to water from nuclear
power plants (accidents and normal operation) and at facilities that produce weapons grade plutonium and
uranium. During the period of 1961–1973, it was estimated that about 514 Ci of ^{137}Cs was emitted to the
Savannah River watershed due to the activities at the Savannah River plutonium processing plant (Olsen
et al. 1989). It was further noted that about 18% of this total (92 Ci) drained directly into the Savannah
River (Olsen et al. 1989). In 1998, it was reported that $1.0x10^{-4}$ Ci of ^{134}Cs and 0.19 Ci of ^{137}Cs were

released in liquid effluents from the Savannah River plutonium processing site in South Carolina (DOE 1998b). In 1993, the NRC estimated that 1.88 Ci of ^{134}Cs and 2.85 Ci of ^{137}Cs were released in liquid effluents from 30 PWR nuclear power plants operating in the United States (NRC 1993a). It was also estimated that 0.12 Ci of ^{134}Cs and 0.58 Ci of ^{137}Cs were released in liquid effluents from 28 BWR nuclear power plants (NRC 1993a). The EPA reported that the total on-site liquid discharge of ^{137}Cs from containment ponds at the Nevada Test Site was 0.0017 Ci in 1997 (EPA 1999c). It was estimated that 1,622 Ci of ^{137}Cs and 811 Ci of ^{134}Cs were released to the cooling pond surrounding the Chernobyl nuclear power plant following the accident in 1986 (UNSCEAR 1996).

Radioactive ^{134}Cs was not identified in water at the three NPL hazardous waste sites where it was detected in some environmental media (HazDat 2003). However, ^{137}Cs was identified in groundwater and surface water at 5 and 3 of the 23 NPL hazardous waste sites, respectively, where it was detected in some environmental media (HazDat 2003).

6.2.3   Soil

Anthropogenic sources of stable ^{133}Cs releases to soils include the mining, milling, and processing of pollucite ore. It is also found in the ash of coal burning power plants and municipal waste incinerators. Stable cesium was detected at concentrations of 0.44–2.01 ppm in the bottom ash of municipal solid waste incinerators operating in the United States (Mumma et al. 1990) and at concentrations of 3–23 ppm from a municipal waste incinerator operating in Barcelona, Spain (Fernandez et al. 1992). Stable cesium was identified in soil at one of the eight NPL hazardous waste sites and in sediment at one of the eight NPL hazardous waste sites where it was detected in some environmental media (HazDat 2003).

Radioactive cesium has been released to soil surfaces by underground nuclear weapons testing, fallout from the accident at the Chernobyl nuclear power plant, and fallout from atmospheric weapons testing. About 1,400 underground tests have been performed worldwide, with a total explosive yield of about 90 megatons (Agency for Toxic Substances and Disease Registry 1999). Small amounts of ^{137}Cs and ^{134}Cs are also released to soil from the normal operation of nuclear power plants and the storage of spent fuel rods. Not including the 30-km exclusionary zone, an area of approximately 2.4×10^4 km^2 near the Chernobyl nuclear power plant was contaminated with ^{137}Cs at a deposition density $>5.4 \times 10^{-5}$ Ci/m^2 following the accident in 1986 (UNSCEAR 1996). Within the exclusionary zone the contamination density may have been 2 orders of magnitude greater in limited areas (UNSCEAR 1996). The mean deposition density of ^{137}Cs and ^{134}Cs in four different soils in Devoke, United Kingdom for May 1986

were reported as $3.7\text{x}10^{-7}$–$5.4\text{x}10^{-7}$ Ci/m^2 and $1.0\text{x}10^{-7}$–$1.8\text{x}10^{-7}$ Ci/m^2, respectively (Hilton et al. 1993). The concentrations of ^{137}Cs in eight sediment cores of the Danube River, Austria were about 540 pCi/kg in April 1985, but increased to approximately 27,000–81,000 pCi/kg in October 1986, following the accident at the Chernobyl nuclear power plant (Rank et al. 1990). The deposition of ^{137}Cs attributed to the accident at the Chernobyl nuclear power plant in sediment at five different sites in Lake Constance, Germany ranged from $2.7\text{x}10^{-7}$ to $2.1\text{x}10^{-6}$ Ci/m^2, while the concentration attributed to fallout from nuclear weapons testing since 1963 ranged from $1.4\text{x}10^{-7}$ to $5.4\text{x}10^{-7}$ Ci/m^2 (Richter et al. 1993). It was estimated that 2,973 Ci of ^{137}Cs and 1,622 Ci of ^{134}Cs were released to the sediments in the cooling pond surrounding the Chernobyl nuclear power plant following the accident in 1986 (UNSCEAR 1996). The deposition density of ^{137}Cs in 123 soil cores collected at the Idaho National Engineering and Environmental Laboratory (INEEL), a site for stored transuranic waste in the United States, ranged from $1.6\text{x}10^{-8}$ to $3.4\text{x}10^{-7}$ Ci/m^2 (DOE 1998a). The deposition density of ^{137}Cs in soils from Idaho, Montana, and Wyoming ranged from $3.0\text{x}10^{-9}$ to $1.1\text{x}10^{-7}$ Ci/m^2, and it was assumed that its origin was fallout from the Nevada Test Site (DOE 1998a). The mean deposition density of ^{137}Cs in the top layer (0–8 cm) of soils near the Chernobyl nuclear power plant in 1988 was $8.6\text{x}10^{-5}$ Ci/m^2 and the mean deposition density of ^{134}Cs was $1.9\text{x}10^{-5}$ Ci/m^2 (Mikhaylovskaya et al. 1993).

It was reported that ^{134}Cs was identified in soil at one site, but was not detected in sediment at the three NPL hazardous waste sites where it was detected in some environmental media (HazDat 2003). It was also reported that ^{137}Cs was identified in soil and sediment at 14 and 5 of the 23 NPL hazardous waste sites, respectively, where it was detected in some environmental media (HazDat 2003).

6.3   ENVIRONMENTAL FATE

6.3.1   Transport and Partitioning

The transport and partitioning of particulate matter in the atmosphere is largely dependent upon the physical properties of the matter such as size and density as well as the meteorological conditions such as temperature, the microphysical structure of the clouds, and rainfall rate. The particle size of ^{137}Cs released to the atmosphere following the Chernobyl nuclear accident was in the range of 0.1–10 μm, (Hirose et al. 1993). Particles <5 μm in diameter usually have low deposition velocities and are transported long distances before being removed from the atmosphere. Atmospheric cesium is deposited on land and water via wet and dry deposition and the deposited cesium may be re-suspended to the atmosphere by disturbances that occur on the ground such as vehicular traffic and construction activity.

The wet deposition velocity of ^{137}Cs at Tsukauba, Japan from May 5 to May 30, 1986 ranged from 0.0026 to 0.110 m/second, and the largest value recorded was during a period of heavy rainfall (Hirose et al. 1993). The mean deposition velocity (wet and dry) of ^{137}Cs measured in Prague, Czechoslovakia was reported as 0.08 m/second from 1989–1992 and the mean flux rate was 1,108 pCi/m^2-year (Rybacek et al. 1994).

Since cesium does not volatilize from water, transport of cesium from water to the atmosphere is not considered likely, except by windblown sea sprays. Most of the cesium released to water will adsorb to suspended solids in the water column and ultimately be deposited in the sediment core. Cesium can also bioconcentrate and has been shown to bioaccumulate in both terrestrial and aquatic food chains. Mean bioconcentration factors (BCF) for ^{137}Cs of 146, 124, and 63 were reported for fish, brown macroalgae, and molluscs, respectively (Fisher et al. 1999). Mean BCF values of 92, 58, 39, and 150 were reported for ^{137}Cs in cod, haddock, plaice, and whiting, respectively (Steele 1990). In a study of aquatic organisms inhabiting the Ottawa River, a 4-fold increase of ^{137}Cs levels was observed with each trophic level (Rowan et al. 1998). The levels of ^{137}Cs in lake trout from Great Slave Lake, Canada were consistently higher than levels found in food sources and a biomagnification factor of 1.9 was calculated for lake trout, relative to their food sources. The biomagnification factor was 3.5 for large mature trout populating the lake (Rowan et al. 1998). It was shown that the bioconcentration and bioaccumulation of ^{137}Cs by aquatic organisms is significantly reduced in waters with a large humic content and high levels of potassium cations (Penttila et al. 1993). Because of the high potassium concentration in oceans, the transfer of ^{137}Cs and ^{134}Cs to fish is much greater in freshwater and the activity of freshwater fish may be 100 times that of ocean fish, given the same cesium concentration in the water (WHO 1983).

In soil surfaces, cesium has low mobility in comparison to other metals and usually does not migrate below a depth of 40 cm. The major portion of cesium is retained in the upper 20 cm of the soil surface (Korobova et al. 1998; Ruse and Peart 2000; Takenaka et al. 1998). Vertical migration patterns of ^{137}Cs in four agricultural soils from southern Chile indicated that approximately 90% of the applied cesium was retained in the top 40 cm of soil, and that in one soil, essentially 100% was bound in the upper 10 cm (Schuller et al. 1997). Migration rates of radiocesium were derived from the depth distribution profiles and were in the range of 0.11 to 0.29 cm/year (Schuller et al. 1997). The vertical migration patterns of ^{90}Sr and ^{137}Cs produced from the atomic bomb exploded in Nagasaki, Japan were studied over a 40-year period (Mahara 1993). Over this period, 95% of the cesium remained in the top 10 cm of the soil surface and no cesium was detected below a depth of 40 cm. In contrast, only 70% of ^{90}Sr was located within a depth of 10 cm and a small percentage was detectable below a depth of 200 cm. The *in situ* vertical

migration rate of ^{90}Sr was calculated as 0.42 cm/year and the migration rate of ^{137}Cs was 0.10 cm/year
(Mahara 1993). The distribution of ^{137}Cs deposited in eastern France due to the Chernobyl accident was
studied recently in four soil types with differing land use. The soils were all high in clay content, organic
matter content never exceeded 10% in any of the soils, and none of the soils had been ploughed or tilled
since the Chernobyl accident in 1986. In all cases, the majority of ^{137}Cs was detected in the upper 10 cm
of soil, with only 1–2% migrating to depths of 40 cm or more (Renaud et al. 2003). Soil adsorption
coefficients (K_d) of five radionuclides (^{54}Mn, ^{60}Co, ^{65}Zn, ^{85}Sr, and ^{137}Cs) were measured for
36 agricultural soils collected in Japan. It was determined that ^{137}Cs had the largest median K_d of all five
radionuclides, and that a positive correlation was observed between the adsorption coefficient and
exchangeable potassium content in the soil (Yasuda et al. 1995). No correlations were observed for other
soil properties such as pH, water content, cation exchange capacity, and exchangeable calcium. Other
studies have reported that clay and zeolite minerals strongly bind cesium cations and can therefore reduce
the bioavailability of cesium and the uptake in plants by irreversibly binding cesium in interlayer
positions of the clay particles (Paasikallio 1999). Experiments conducted by growing plants in a peat soil
showed that the introduction of zeolites into the soil-plant system decreased the uptake of ^{134}Cs in plants
by a factor of 8 (Shenber and Johanson 1992). The low hydration energy of cesium cations is primarily
responsible for their selective sorption and fixation by clays and zeolites (Hakem et al. 1997). Soils rich
in organic matter adsorb cesium, but the cesium adsorbed in the organic fraction is readily exchangeable
and highly available for plant uptake (Sanchez et al. 1999). Regions in Venezuela, Brazil, and Russia
have been identified where a great deal of rain is encountered, the soil is peaty or podzolic (a type of
forest soil characterized by high leachability), and the mobility of cesium is considerably greater than in
other soils (LaBrecque and Rosales 1996; WHO 1983).

The plant/soil concentration ratio (activity/kg of plant/activity/kg of soil) of ^{137}Cs for field crops in
southern Finland ranged from 0.01 to 0.26. In northern Finland, this ratio ranged from 0.01 to 2.29, with
the lowest values occurring in clay and silt soils and the highest values observed in fine sands (Paasikallio
et al. 1994). The plant/soil concentration ratios for a series of vegetables and grains decreased in the
following order: lettuce, cabbage>carrot, potato>cereals, onion; for fruits, the order was: black-
currant>strawberry>apple (Paasikallio et al. 1994). The mean plant/soil concentration ratios of ^{137}Cs for
trees at the Hanford Waste Site in the United States were 0.03 (roots), 0.06 (cores), and 0.02 (leaf/twig)
(Landeen and Mitchell 1986).

6.3.2   Transformation and Degradation

6.3.2.1   Air

When pure cesium metal is exposed to air, an explosion-like oxidation occurs, forming a mixture of cesium oxides (Cs_2O, Cs_2O_2, and Cs_2O_3). Cesium compounds released to the atmosphere will eventually settle to earth by wet and dry deposition. Radioactive forms of cesium such as ^{137}Cs and ^{134}Cs are continuously transformed to stable isotopes of barium or xenon by the natural process of radioactive decay. The pathways and mechanisms of these reactions have been described in Chapter 4.

6.3.2.2   Water

When pure cesium metal is released to water, a vigorous reaction occurs yielding cesium hydroxide (CsOH), the strongest base known, and hydrogen gas, which may ignite spontaneously. In general, cesium compounds are very water soluble, in which cesium is present in the ionic form as the Cs^+ cation. Under normal environmental conditions, Cs^+ cations are neither degraded nor transformed, but may adsorb to suspended solids and sediment in the water column, forming insoluble complexes.

6.3.2.3   Sediment and Soil

Cesium salts and most cesium compounds are generally very water soluble, with the exception of cesium alkyl and aryl compounds, which have low water solubility. Cesium cations have low hydration energy and can react with clay minerals, zeolites, or soils with a high percentage of exchangeable potassium, forming insoluble, immobile complexes.

6.4   LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to cesium depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. Radioactive cesium is measured in units of activity, not mass. A great deal of monitoring data for radioactive cesium in environmental samples prior to, and shortly following, the accident at the Chernobyl nuclear power plant on April 26, 1986 has been included in this chapter.

6.4.1   Air

Data reporting the background levels of [133]Cs in the atmosphere are limited. Since the amount of cesium mined and milled is small, it is expected that background concentrations in the atmosphere will be low. The concentration of [133]Cs measured in the South Pole during 1974–1975 was reported in the range of 0.072–0.14 pg/m^3, with a mean of 0.10 pg/m^3 (Maenhaut et al. 1979). The maximum airborne concentrations of [133]Cs at the Tera Nova Bay Station in the Arctic were reported as 20–90 pg/m^3 during the 1989–1990 Italian expedition and 10–60 pg/m^3 for the 1990–1991 expedition (Chiavarini et al. 1994). The average concentration of [133]Cs in precipitation collected in Lennox, Massachusetts during the summer of 1984 was reported as 0.0075 µg/L, with a range from 0.0050 to 0.024 µg/L (Dasch and Wolff 1989).

Radioactive [134]Cs and ^{137}Cs have been detected at various concentrations (activities) in the atmosphere following the accident at the Chernobyl nuclear power plant on April 26, 1986. The greatest concentrations were observed in locations throughout Russia and Europe, but detectable levels were measured globally, including in North America. The air concentrations of [134]Cs above the destroyed reactor were 1,756.8 pCi/m^3 on May 8 and 54.0 pCi/m^3 on May 18 (Buzulukov and Dobrynin 1993). The concentrations of ^{137}Cs in the air above the destroyed reactor were 8,918.9 pCi/m^3 on May 8 and 135.1 pCi/m^3 on May 18 (Buzulukov and Dobrynin 1993). Atmospheric concentrations of [134]Cs in Belgrade, Yugoslavia were 5.4×10^{-4} (May 1, 1986), 160.5 (May 2, 1986), 145.0 (May 3, 1986), and 25.4 (May 4, 1986) pCi/m^3, while atmospheric concentrations of ^{137}Cs were 0.001 (May 1, 1986), 324 (May 2, 1986), 276 (May 3, 1986), and 49 (May 4, 1986) pCi/m^3 (Ajdacic and Martic 1990). Cesium concentrations decreased significantly over time as advection, wet deposition, and dry deposition lowered the atmospheric concentrations. The average monthly atmospheric concentrations of ^{137}Cs in Belgrade that were measured during 1991–1996 ranged from 1.3×10^{-4} to 2.0×10^{-3} pCi/m^3, with a pronounced maximum during the winter months (Todorovic et al. 1999). Monitoring data from Prague, Czechoslovakia indicated that there was a gradual decrease in the atmospheric concentration of ^{137}Cs from May 1986 to February 1992. Maximum concentrations were observed immediately following the accident at the Chernobyl nuclear power plant (approximately 0.05 pCi/m^3) and gradually decreased over a 6-year period (Rybacek et al. 1994). The atmospheric concentration of [134]Cs decreased more rapidly, presumably due to the shorter half-life of [134]Cs. The concentration of ^{137}Cs in the atmosphere of Thessaloniki, Greece ranged from 8.1×10^{-4} to 0.044 pCi/m^3 from July 1987 to December 1988, and the concentration of ^{137}Cs in rainfall ranged from 0.27 to 34 pCi/L from November 1986 to February 1989 (Papastefanou et al. 1989). During a heavy rainfall event occurring on May 6, 1986, ^{137}Cs was detected at a concentration of 46,000 pCi/L (Papastefanou et al. 1989). The concentration of ^{137}Cs in Tsukuba, Japan

during May 1986 ranged from about 0.054 to 1.6 pCi/m^3 (Hirose et al. 1993). The average atmospheric

concentrations of ^{134}Cs and ^{137}Cs in eastern Canada were reported as 0.024 and 0.046 pCi/m^3,

respectively, during May 10–24, 1986 (Huda et al. 1988). The maximum atmospheric concentration of

^{137}Cs measured in New York City in May 1986 was reported as 0.26 pCi/m^3 (Feely et al. 1988). The

average concentrations of ^{137}Cs and ^{134}Cs in Barrow, Alaska were reported as 0.027 and 0.019 pCi/m^3,

respectively, for the month of May 1986 (DOE 1986).

6.4.2   Water

The background concentration of ^{133}Cs in fresh water lakes and rivers is ordinarily in the range of 0.01 to

1.2 µg/L, and the concentration in seawater is about 0.5 µg/L (WHO 1983). Stable cesium was detected

in streams that feed into the Tamagawa River, Japan at concentrations of 9x10^{-4} to 0.093 µg/L (Tanizaki

et al. 1992). Studies from five distinct geochemical areas of the semi-arid endorheic region of the

Famatina Range (La Rioja, Argentina) have shown some areas contain high levels of ^{133}Cs in natural

waters and sediment (Fernandez-Turiel et al. 1995). The cesium concentration in fresh water systems of

this region ranged from 0.58 to 3.69 µg/L (Fernandez-Turiel et al. 1995). The concentrations of 19 trace

metals were studied in drinking water and groundwater supplies in southern Nigeria. Stable cesium was

detected in groundwater and drinking water at mean concentrations of 0.61 µg/L (range 0.09–3.72 µg/L)

and 0.35 µg/L (range 0.05–4.32 µg/L), respectively (Asubiojo et al. 1997). The mean concentration of

cesium in drinking water was lower than the mean concentration of any of the other trace elements

analyzed.

High and low level radioactive wastes were dumped by the former Soviet Union into remote Arctic

waters, leading to the release of radioactive cesium into the Kara and Barents Seas. The level of ^{137}Cs in

surface water of the Barents Sea and Kara Sea was 0.14 and 0.16 pCi/L, respectively, and ^{137}Cs was also

detected in deep water of the Barents Sea at a concentration of 0.15 pCi/L (Fisher et al. 1999). The

concentration of ^{137}Cs in the Black Sea ranged from 2.7 to 8.1 pCi/L for the period of 1991–1996, with

the exception of the spring of 1992, when concentrations as high as 43 pCi/L were observed (Strezov et

al. 1999). Preliminary measurements taken in the vicinity of the nuclear submarine, the Kursk, which

accidentally sank in the Barents Sea in August 2000, indicate that that no leakage of radionuclides from

the reactor has occurred and only background levels of radioactive cesium (and other radionuclides) have

been observed in water and sediment near the submarine (Amundsen et al. 2002). From 1988 to 1991, the

mean concentrations of ^{137}Cs and ^{134}Cs along the Spanish coast of the Mediterranean Sea were 0.13 and

0.0072 pCi/L, respectively (Molero et al. 1999). Due to its shorter half-life, ^{134}Cs was detected in all

14 samples collected in 1988 and 1989, but only in 3 samples collected in 1990 and 1991, suggesting that

the ^{134}Cs levels observed in surface Mediterranean waters during this period were due mainly to

Chernobyl-related deposition. The ^{137}Cs concentration incorporated into the Mediterranean Sea near the

Spanish coast from the post-Chernobyl fallout was about 0.032 pCi/L, which was approximately a 33%

increase over previous levels (Molero et al. 1999). Maximum ^{137}Cs and ^{134}Cs levels in the immediate

vicinity of nuclear power plants located on the southern Catalan shore of the Mediterranean were 0.57 and

0.059 pCi/L (Molero et al. 1999). Concentrations of ^{137}Cs in lakes and streams in Devoke, United

Kingdom decreased exponentially from a maximum concentration of about 8.1 pCi/L on May 6, 1986 to

about 0.027 pCi/L 1,200 days later (Hilton et al. 1993). The mean concentration of ^{137}Cs in six lakes

located in central Finland ranged from 111 pCi/L in 1987 to 8.1 pCi/L in 1989 (Penttila et al. 1993).

The concentration of ^{137}Cs and ^{134}Cs in groundwater at 18 U.S. Department of Energy (DOE) facilities

was reported in the range of 2.7×10^{-3} to 1.83×10^{3} pCi/L (DOE 1992). The concentration of ^{137}Cs

measured in groundwater wells at Carlsbad, New Mexico (the site of Project GNOME) ranged from 99 to

6.8×10^{5} pCi/L in 1997 (EPA 1999c). The concentration of ^{137}Cs in groundwater at the Chernobyl nuclear

power plant was in the range of 40.5–1,100 pCi/L in 1988 and 29.7–129.7 pCi/L in 1989 (Prister et al.

1990). The mean concentration of ^{137}Cs in drinking water in Poland in 1974 was reported as 0.2 pCi/L

(Glowiak et al. 1977b). These concentrations in water may be compared to the federal radiation safety

standards. For continuous ingestion over a lifetime, the maximum concentrations of ^{134}Cs and ^{137}Cs in

drinking water are limited to 900 and 1,000 pCi/L respectively; these limits assume no other intake of

radioactivity. If other radioisotopes are ingested, then the intake limits for each must be adjusted

proportionately (USNRC 1999a).

6.4.3   Sediment and Soil

Stable cesium occurs naturally in the earth's crust at low concentrations. Granites contain an average

cesium concentration of about 1 ppm and sedimentary rocks contain about 4 ppm (Burt 1993). Others

have estimated cesium concentration of granites as high as 5 ppm (WHO 1983). Stable cesium was

detected in dust samples from roadside and pedestrian traffic in Nagpur, India at concentrations of 1.53–

3.63 µg/g, with the largest value obtained in the vicinity of a metals factory (Chutke et al. 1995). ^{133}Cs

was detected at concentrations ranging from 0.9 to 2.2 µg/g in alluvial sediments in the Sava River,

Croatia (Vertacnik et al. 1997). The concentration of ^{133}Cs in river sediment from five distinct

geochemical areas of the semi-arid endorheic region of the Famatina Range (La Rioja, Argentina) was

2.28–6.20 µg/g (Fernandez-Turiel et al. 1995).

The concentration of ^{137}Cs in soils of Thessaloniki, Greece ranged from 1,440 to 35,324 pCi/kg (average 8,154 pCi/kg) and the concentration of ^{134}Cs ranged from about 270 to 5,676 pCi/kg during the period of August 1986 to February 1989, with most of the fallout attributed to the accident at the Chernobyl nuclear power plant (Papastefanou et al. 1989). The concentration of ^{137}Cs in 10 uncultivated fields from southern England ranged from 0 to 946 pCi/kg, with the highest levels contained in the top 10 cm of the soil surface (Owens et al. 1996). The concentration of ^{137}Cs in five cultivated fields ranged from 0 to 540 pCi/kg, and the concentrations were well distributed from the surface to the plough layer (Owens et al. 1996). The concentration of ^{137}Cs in three soils in Hong Kong receiving a large amount of rainfall ranged from 32 to 201 pCi/kg (Ruse and Peart 2000). The concentration of ^{137}Cs in sediment from the Ribble Estuary, England near the British Nuclear Fuels Laboratory ranged from 270 to 1,351 pCi/kg (Brown et al. 1999). The average concentration of ^{137}Cs in sediment from Lake Ontario was reported as 8,108 pCi/kg, and suspended solids from the Niagara River contained 324 pCi/kg, measured during the 1983–1986 sampling period (Platford and Joshi 1989). The average concentrations of ^{137}Cs in uncultivated soils in northern Poland were reported as 616–4,170 pCi/kg from 1988 to 1991 (Pietrzak-Flis et al. 1994). The mean concentration of ^{137}Cs in surface soil samples from the Los Alamos nuclear laboratory test site during the period of 1974–1996 was reported as 611 pCi/kg (Fresquez et al. 1998). Concentrations of ^{137}Cs around the perimeter of the site and background concentrations off the site were reported as 589 and 419 pCi/kg , respectively. The concentration of ^{137}Cs and ^{134}Cs in soils and sediments at 18 U.S. DOE facilities was reported to range from 20 to 4.69×10^{7} pCi/kg (DOE 1992). The concentration of ^{137}Cs in sediment from the Savannah River ranged from 5 to 100 pCi/kg in 1986, while the concentration in suspended solids and particulate matter ranged from 240 to 4.324×10^{6} pCi/kg (Olsen et al. 1989). The mean concentration of ^{137}Cs in soil at the Hanford Site in the United States was 4,540 pCi/kg (Landeen and Mitchell 1986). The mean concentration of ^{137}Cs in soils taken from two high-elevation sites in northern Colorado ranged from 4,054 to 7,027 pCi/kg (Ulsh et al. 2000).

The concentration of ^{137}Cs in sediment near the nuclear submarine, the Kursk, which accidentally sank in the Barents Sea in August 2000, ranged from about 18.9 to 40.5 pCi/kg (Amundesen et al. 2002). These levels are considered typical background levels in the Barents Sea and are attributed to releases from the Sellafield reprocessing facility in the United Kingdom, Chernobyl related fallout, and deposition from past nuclear weapons tests.

6.4.4   Other Environmental Media

Information on the concentrations of ^{137}Cs, ^{134}Cs, and ^{133}Cs in various animals is shown in Table 6-2. Concentrations are dependent on the location, date, and level of exposure. For example, the concentrations of ^{137}Cs in bullhead catfish inhabiting an abandoned nuclear reactor reservoir at the Savannah River site in South Carolina were as high as 1.54×10^5 pCi/kg (McCreedy et al. 1997), but concentrations in various freshwater species of fish in the Ottawa River were in the range of 54 to 351 pCi/kg (Rowan et al. 1998). After the accident at the Chernobyl nuclear power plant, the average concentrations of ^{137}Cs in perch, pike, and roach obtained from 52 freshwater lakes in Finland were 55,811, 66,297, and 15,270 pCi/kg, respectively, in 1988. By contrast, the mean concentrations in 1992 had fallen to 14,324, 18,567, and 5,892 pCi/kg, respectively (Sarkka et al. 1996). The concentration of total ^{137}Cs and ^{134}Cs measured in body tissue from sheep in Ireland from 1989 to 1991 ranged from about 2,700 to 10,811 pCi/kg, with highest levels observed during November of 1989 (McGee et al. 1993).

The concentrations of ^{133}Cs in plants from Britain were studied over a 1-year period. Concentrations of 50–300 ng/g were measured, with the highest levels observed during the summer and fall months. The median concentration of ^{133}Cs in poplar leaves collected in Bulgaria was reported as 75 ng/g, while concentrations in land plants ranged from 30 to 440 ng/g (Djingova et al. 1995). In contrast to the radioactive isotopes of cesium, concentrations of stable ^{133}Cs depend primarily on the root uptake from soil and not on atmospheric deposition. Lichens and mosses have been shown to trap and retain ^{137}Cs and ^{134}Cs more than vascular plants due to their relatively large surface area. Lichens and mosses from northern Greece contained ^{137}Cs levels of 6.6×10^4–5.1×10^5 pCi/kg during the period of 1989–1991 (Papastefanou et al. 1992) and moss samples from Finland collected in 1988–1989 contained 4.3×10^4–9.7×10^5 pCi/kg (Penttila et al. 1993). The mean concentration of ^{137}Cs in three species of lichens collected in August 1986 from Megalopolis, Greece were 2.6×10^4–3.3×10^4 pCi/kg, while the mean concentrations for the same three species of lichens collected in October 1996 had fallen to 3,324–7,892 pCi/kg (Riga-Karandinos and Karandinos 1998). Mushrooms, lichens, and mosses obtained near Manitoba, Canada in August 1986 contained ^{137}Cs at mean concentrations of 6.4×10^5, 8.6×10^4, and 8.4×10^4 pCi/kg, respectively (Mihok et al. 1989). Since caribou and reindeer consume large amounts of this vegetation during the winter months, elevated levels of ^{137}Cs and ^{134}Cs have been detected in these animals.

Stable cesium has been detected infrequently in food products at low concentrations. The average concentration of stable ^{133}Cs in 110 onion samples collected in Denmark was 0.21 µg/kg, with a range of

Table 6-2. Concentration of ^{133}Cs, ^{134}Cs, and ^{137}Cs in Animals

Species	Location and date	^{133}Cs (µg/g)	^{134}Cs (Bq/kg)	^{137}Cs (Bq/kg)	Source
Royal albatross (N=3)	Indian Ocean (1994)	0.009–0.025 (0.014 mean); liver	No data	No data	Kim et al. 1998
Black-footed albatross (N=18)	North Pacific (1985)	0.007–0.049 (0.022 mean); liver	No data	No data	Kim et al. 1998
Black-browed albatross (N=9)	Indian Ocean (1994)	0.013–0.042 (0.022 mean); liver	No data	No data	Kim et al. 1998
White-capped albatross (N=3)	Indian Ocean (1994)	0.011–0.039 (0.029 mean); liver	No data	No data	Kim et al. 1998
Yellow-nosed albatross (N=4)	Indian Ocean (1994)	0.0029–0.079 (0.022 mean); liver	No data	No data	Kim et al. 1998
Grey-headed albatross (N=10)	Indian Ocean (1994)	0.011–0.031 (0.016 mean); liver	No data	No data	Kim et al. 1998
Northern giant petrel (N=6)	Indian Ocean (1994)	0.005–0.034 (0.015 mean); liver	No data	No data	Kim et al. 1998
Northern fulmar (N=17)	North Pacific (1985)	0.008–0.036 (0.016 mean); liver	No data	No data	Kim et al. 1998
Bluefin tuna (N=14)	Newfoundland (1990)	0.08–0.24 (0.13 mean); muscle	No data	No data	Hellou et al. 1992a
Cod (N=12)	Newfoundland (1990)	0.16–0.24 (0.19 mean); muscle	No data	No data	Hellou et al. 1992b
Cod (N=10)	Newfoundland (1990)	0.14–0.36 (0.23 mean); muscle	No data	No data	Hellou et al. 1992b
Pilot whale (N=9)	North Atlantic (1987–1996)	0–0.010 (0.006 mean); liver	No data	No data	Becker et al. 1997
White-sided dolphin (N=4)	North Atlantic (1987–1996)	0.027–0.042 (0.032 mean); liver	No data	No data	Becker et al. 1997
Beluga whale (N=15)	Arctic (1987–1996)	0.021–0.046 (0.031 mean); liver	No data	No data	Becker et al. 1997
Ringed seal (N=13)	Arctic (1987–1996)	0.0045–0.048 (0.020 mean); liver	No data	No data	Becker et al. 1997
Woodcock (N=24)	Ireland (1986)	No data	3.9–206.4; muscle	6.2–565.5; muscle	Pearce 1995
Duck (N=5)	Ireland (1986)	No data	2.2–14.3; muscle	6.4–18.0; muscle	Pearce 1995
Snipe (N=5)	Ireland (1986–1987)	No data	1.0–5.4; muscle	3.6–16.9; muscle	Pearce 1995
Reindeer (N=8)	Northern Sweden (1986–1987)	No data	No data	900 (mean); muscle	Ahman and Ahman 1994
Deer (N=11)	Los Alamos (1991–1998)	No data	No data	2,516 (mean); muscle	Fresquez et al. 1999a

Table 6-2. Concentration of ^{133}Cs, ^{134}Cs, and ^{137}Cs in Animals

Species	Location and date	^{133}Cs (µg/g)	^{134}Cs (Bq/kg)	^{137}Cs (Bq/kg)	Source
Deer (N=11)	Los Alamos (1991–1998)	No data	No data	888 (mean); bone	Fresquez et al. 1999a
Caribou (N=18)	Saskatchewan (1995)	No data	No data	58 (mean); bone	Thomas and Gates 1999
Caribou (N=18)	Saskatchewan (1995)	No data	No data	228 (mean); liver	Thomas and Gates 1999
Caribou (N=18)	Saskatchewan (1995)	No data	No data	367 (mean); muscle	Thomas and Gates 1999
Caribou (N=18)	Saskatchewan (1995)	No data	No data	553 (mean); kidney	Thomas and Gates 1999
Caribou (N=36)	Alaska (1987)	No data	No data	26–232; neck	Allaye-Chan et al. 1990
Caribou (N=36)	Alaska (1987)	No data	No data	28.4–201.1; shoulder	Allaye-Chan et al. 1990
Caribou (N=36)	Alaska (1987)	No data	No data	30.2–166.5; back	Allaye-Chan et al. 1990

not detected to 0.98 μg/kg (Bibak et al. 1998). By comparison, other elements such as calcium and potassium had mean concentrations of $2x10^5$ and $1.6x10^6$ μg/kg, respectively. The concentration range of ^{133}Cs in wheat flour samples collected in Pakistan was 6.7–11.2 ppb, but ^{133}Cs was not detected from wheat flour samples from America (Ahman et al. 1994).

Levels of ^{137}Cs were below detection limits for all foods analyzed for in the U.S. Food and Drug Administration (FDA) Total Diet Study in 1991–1996 with the exception of honey (Capar and Cunningham 2000). The concentration of ^{137}Cs in honey from the 1995 Market Basket Survey was 6.7 Bq/kg (181.1 pCi/kg), which is almost 200 times lower than the regulatory level of ^{137}Cs in foods. The average concentrations of total ^{137}Cs and ^{134}Cs in milk powder, infant milk powder, infant cereal, meat, lentil, wheat, and macaroni samples from Saudi Arabia were 514, 351, 486, 162, 270, 676, and 351 pCi/kg, respectively (Abdul-Majid et al. 1992). For the month of June 1986, the average concentrations of total ^{137}Cs and ^{134}Cs in milk, green vegetables, fruit, lamb, and beef were reported as 3,243, 2,703, 2,703, 8,108, and 1,622 pCi/kg in high deposition areas of the United Kingdom (Cumbria, north Wales, Scotland, northern Ireland, and the Isle of Man) (Mondon and Walters 1990). The concentration of total ^{137}Cs and ^{134}Cs was <676 pCi/kg in each of these food sources in areas of low deposition during this time frame. The maximum concentration of ^{137}Cs in pasteurized milk from 65 cities in the United States was 14 pCi/L in May 1989 (EPA 1989). The concentration of ^{137}Cs in fresh milk from Chester, New York and pasteurized milk samples from New York City in May 1986 ranged from about 5.4 to 18.9 pCi/L (Feely et al. 1988). Using radiological surveys from 1978 and 1985–1986, the concentration of ^{137}Cs in 44 adult food groups from the Rongelap Island and Rongelap Atoll was in the range of 0.52–13,000 pCi/kg (Robinson and Phillips 1989).

6.5   GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

General population exposure to cesium occurs by ingestion of food and drinking water, by inhalation of ambient air, and dermal contact with cesium in soil. Since stable ^{133}Cs is found in low concentrations in the environment, the exposure to the general population is also low. The National Health And Nutritional Examination Survey (NHANES), conducted by the Centers for Disease Control and Prevention (CDC), reported that the geometric mean concentration of ^{133}Cs in the urine of 2,464 U.S. residents in various population groups was as follows: children aged 6–11 years, 4.87 μg/L; adolescents aged 12–19 years, 4.54 μg/L; adults aged 20 or older, 4.26 μg/L (CDC 2003). Based upon gender, males had a higher geometric mean concentration (4.83 μg/L), as compared to females (3.94 μg/L). Occupational exposure to ^{133}Cs occurs primarily through inhalation and dermal contact at workplaces where pollucite is mined or

cesium compounds are manufactured or used. The National Occupational Exposure Survey (NOES) conducted by the National Institute for Occupational Safety and Health (NIOSH) from 1981 to 1983 has estimated that 16,461 workers (4,276 of these were female) were potentially exposed to cesium and cesium compounds in the United States (NOES 1989). The NOES database does not contain information on the frequency, level, or duration of the exposure of workers to any of the chemicals listed therein. They are surveys that only provide estimates of workers potentially exposed to the chemicals. The median concentration of ^{133}Cs in the lungs of metal workers in northern Sweden was 0.016 µg/g and was lower than the median concentration of 0.021 µg/g for a control group that was not occupationally exposed (Hewitt 1988). The range of ^{133}Cs concentrations in lung tissue of coal miners from the United Kingdom was 0.07–0.91 µg/g (Hewitt 1988).

Exposure to radioactive cesium is of greater concern from a health perspective than exposure to stable cesium. The primary source of radioactive cesium in the environment is due to fallout from past atmospheric nuclear weapons tests and the Chernobyl accident. Additional contributions from the normal operation of nuclear power plants and other nuclear facilities are small by comparison. Current exposure of the general U.S. population to ^{134}Cs and ^{137}Cs is expected to be low since atmospheric testing of nuclear weapons was discontinued in 1963 and Chernobyl-related fallout was low in the United States. As discussed in Appendix A, the average annual effective dose of ionizing radiation (including ^{134}Cs and ^{137}Cs) to the U.S. population from anthropogenic sources is negligible in comparison to natural sources, especially radon and its decay products.

The average daily intake (AVDI) of ^{137}Cs and ^{134}Cs was estimated for adult males residing in the Ukraine in 1994, based upon total diet samples. The mean intake of ^{137}Cs was estimated as 109 pCi/day and the mean intake of ^{134}Cs was estimated as 8.1 pCi/day (Shiraishi et al. 1997). Based on dietary patterns and the concentration of radiocesium in food sources, the total dietary intakes of ^{134}Cs and ^{137}Cs for typical adults residing in Croatia for the month of May 1986 were estimated as $2.8x10^4$ and $5.9x10^4$ pCi, respectively (Lokobauer et al. 1988). The mean monthly levels of ^{137}Cs in human muscle tissue in Graz, Austria were reported as 1,519 (July 1986 to June 1987), 1,049 (July 1987 to June 1988), 340.5 (July 1988 to June 1989), and 202.7 pCi/kg (July 1989 to June 1990), with a maximum value of 9,584 pCi/kg observed in an individual during September 1986 (Rabitsch et al. 1991). The monthly averages for ^{134}Cs were about half of those reported for ^{137}Cs. By comparison, the maximum concentration of ^{137}Cs in muscle tissue from Harwell, England in 1959 was reported as 224 pCi/kg, the mean concentration in muscle tissue from Massachusetts during January 1961 to June 1962 was 100 pCi/kg, and the mean concentration in human muscle tissue obtained from Japan during April to December 1963 was reported

as 119 pCi/kg (Rabitsch et al. 1991). The mean concentrations of ^{137}Cs in the urinary excretion of Italians in northern Italy were 7.3 and 6.2 pCi/day in 1995 and 1996, respectively (Ropolo and Cesana 1997). These values were about two orders of magnitude less than values reported for measurements taken in 1987. The mean concentration of ^{137}Cs in brain, heart, liver, gonads, muscle, bone, and teeth were 0.440, 1.860, 0.490, 2.440, 0.017, 0.106, and 0.23 pCi/g, respectively, for adult cadavers over 34 years of age in Poland during 1975 (Glowiak et al. 1977a). The mean body burdens of ^{137}Cs for adults in Helsinki, Viitasaari, and Ammans, Finland from 1987 to 1994 are given in Table 6-3 (Rahola and Suomela 1998).

Persons employed at nuclear power facilities and waste disposal sites are potentially exposed to higher levels of ^{137}Cs and ^{134}Cs than the is general population. The NOES conducted by NIOSH from 1981 to 1983 estimated that 13,148 workers (1,294 of these were female) were potentially exposed to ^{134}Cs and ^{137}Cs in the United States (NOES 1989).

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 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 1993a).

As for adults in the general population, exposures of children to cesium occur from normal ingestion of food and drinking water, inhaling air, and dermal contact with cesium in soil. No information on cesium levels in amniotic fluid, meconium, cord blood, or neonatal blood was available.

Radioactive cesium was detected in several pasteurized milk and breast milk samples worldwide, and since children tend to consume large amounts of milk, this represents an important source of childhood

Table 6-3. The Mean Body Burdens of ^{137}Cs for Adults in Finland from 1987 to 1994[a]

Location	Date	Concentration (pCi/kg)
Helsinki, Finland	1987	649
	1988	568
	1989	405
	1990	297
	1991	246
	1992	214
	1993	195
	1994	181
Viitasaari, Finland	1987	3,514
	1988	1,946
	1989	1,595
	1990	1,081
	1991	838
	1992	676
	1993	649
	1994	514
Ammans, Finland	1987	2,892
	1988	3,108
	1989	2,243
	1990	2,486
	1991	1,568
	1992	1,405
	1993	865
	1994	811

[a]Rahola and Suomela 1998

exposure. The maximum concentration of ^{137}Cs in pasteurized milk from 65 cities in the United States was 14 pCi/L in May 1989 (EPA 1989). Concentrations of ^{137}Cs in human milk samples from several U.S. cities from 1956 to 1961 were <20 pCi/L (Eaman 1986). The overall ^{137}Cs concentrations in milk taken 7 days postpartum from 37 mothers in two Italian hospitals were 5.8–115 pCi/L (Eaman 1986). In a study of females from northern Sweden, ^{137}Cs was detected in breast milk from 8 out of 12 mothers at concentrations of 7.3–178.4 pCi/kg (Johansson et al. 1998). The infants of these mothers who were breast-fed had whole-body levels of ^{137}Cs in the range of 45.9–675.7 pCi/kg (Johansson et al. 1998). Based on dietary patterns and the concentration of radiocesium in food sources, the total dietary intakes of ^{134}Cs and ^{137}Cs for children (10 years old) residing in Croatia for May–June 1986 were estimated as 43,000 and 190,000 pCi, respectively (Lokobauer et al. 1988). For infants (1 year of age), it was estimated that the total intakes of ^{134}Cs and ^{137}Cs were 46,000 and 170,000 pCi, respectively (Lokobauer et al. 1988). The total intakes of ^{134}Cs and ^{137}Cs for adults during this same time period were estimated as 40,000 and 84,000 pCi, respectively (Lokobauer et al. 1988). The higher intakes of ^{134}Cs and ^{137}Cs for infants and children were linked to a greater consumption of contaminated milk.

The tendency of young children to ingest soil, either intentionally through pica or unintentionally through hand-to-mouth activity, is well documented. These behavioral traits can result in ingestion of cesium present in soil and dust. Soil ingestion may be a potentially important exposure pathway in areas that have historically had a great deal of ^{134}Cs and ^{137}Cs deposited onto soil surfaces from the accident at the Chernobyl nuclear power plant or fallout from weapons testing. Playing in contaminated soil could also lead to dermal and external exposure. Ingested cesium is adsorbed strongly to soils and may not be in bioavailable form. In a study in which 102 healthy volunteers were fed ^{134}Cs-contaminated soil pellets, only about 1% of the original amount was absorbed and on average, about 60% of the intake was excreted within 48 hours (LeRoy et al. 1966).

It is unlikely that children whose parents are employed at nuclear power generating plants and facilities that store or handle radioactive waste are exposed to ^{134}Cs and ^{137}Cs from parents' clothing or items removed from the workplace because workers pass through exit monitors at nuclear power plants, and the extensive use of outerware that remains in the plant prevents these types of incidents. It is also unlikely that children are exposed to ^{133}Cs from parents' clothing or items that have been removed from the workplace if the parents are employed in the mining, milling, or processing of pollucite ore. Other home exposures are unlikely since household items or products used in crafts, hobbies, or cottage industries do not contain significant amounts of cesium.

6.7   POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Few populations are likely to be exposed to high levels of ^{133}Cs. Persons residing in the vicinity of pollucite mines and workers employed in the mining, milling, and production of cesium may be exposed to higher levels than the general population.

Human exposure to ^{134}Cs and ^{137}Cs can be external, due to exposure from a radioactive cloud and contaminated environmental media after deposition, or internal, via inhalation and ingestion of contaminated food or drinking water. In both cases, populations residing near the source are exposed to potentially higher doses than populations far removed from the source. Humans residing near areas where nuclear weapon testing was previously conducted may have been exposed to higher doses of radiation from ^{134}Cs and ^{137}Cs, both internally and externally, than the general population. Populations residing in southern Utah and Nevada were exposed to radioactive cesium (and many other radionuclides) due to testing conducted at the Nevada Test Site (NTS). A total of 100 surface or near-surface tests with a total explosive yield of about 1 megaton were performed at the Nevada test site between 1951 and 1962. The dust from these tests also drifted over the continental United States, producing varying degrees of exposure for remote populations depending upon the meteorological conditions. For example, the greatest non-local fallout from one of the tests was received in New York State in 1953, some 2,000 miles away from the source, due to wet deposition (Agency for Toxic Substances Disease Registry 1999). About 500 underground tests were also performed at the NTS, but underground testing rarely leads to significant off-site contamination (Agency for Toxic Substances Disease Registry 1999). The EPA Office of Radiation and Indoor Air conducts off-site environmental monitoring around former U.S. nuclear test areas. The 1997 report concluded that the current exposure to populations around the NTS from radionuclides, including ^{137}Cs and ^{134}Cs, was negligible in comparison to background levels (EPA 1999c).

Populations residing in the vicinity of nuclear power plants may also be exposed to higher levels of ^{134}Cs and ^{137}Cs than the general population due to airborne and liquid effluents from these plants. Persons employed in these facilities are also likely to be exposed to greater levels than the general population, particularly those employees who must handle radioactive waste material. However, despite the potential for exposure, increased body burdens of radioactive cesium have not been observed among the population of workers in nuclear facilities or among members of the off-site general public.

Human populations that received a large amount of fallout from the Chernobyl nuclear accident were potentially exposed to high levels of radiocesium. These populations were primarily located in the Ukraine and northern Europe that received a great deal of rainfall in the weeks following the Chernobyl

accident. Not including the 30-km exclusionary zone, an area of approximately 2.4×10^4 km^2 near the
plant was contaminated with ^{137}Cs at a deposition density $>5.4 \times 10^{-5}$ Ci/m^2 (UNSCEAR 1996). Within the
exclusionary zone, the contamination density may have been a factor of about 100 greater in limited areas
(UNSCEAR 1996). The Bryansk-Belarus region about 200 km northeast of the reactor had deposition
densities as high as 1.3×10^{-4} Ci/m^2, and the Kaluga-Tula-Orel location approximately 500 km northeast of
the reactor had deposition densities of about 1.6×10^{-5} Ci/m^2 (Agency for Toxic Substances Disease
Registry 1999). While cesium is not considerably taken up by the roots of vascular plants, the deposition
of radioactive debris on flora with large surface areas such as lichens or moss is significant (see
Section 6.4.4). As a result, animals that feed on this vegetation such as caribou or reindeer may ingest
large quantities of radiocesium. Nordic or Eskimo populations that use these animals as an important
source for food are exposed to larger quantities of ^{137}Cs and ^{134}Cs than the general population (Allaye-
Chan et al. 1990; WHO 1983). The average concentration of ^{137}Cs in the skeletal muscle of a herd of
caribou from Alaska was in the range of 76–133 Bq/kg in 1987 (Allaye-Chan et al. 1990). According to
the National Council on Radiation Protection and Measurements, the maximum nonoccupational
radiocesium intake is 300,666 Bq/year (8.1×10^6 pCi/year) for adults (NCRP 1977). At the maximum
average skeletal muscle concentration (133 Bq/kg), an annual consumption of over 2,260 kg of caribou
meat would be required to reach this limit (assuming no other intake sources). Using the mean ^{137}Cs level
of 900 Bq/kg in the muscle of reindeer obtained from northern Sweden in 1986–1987 (Ahman and
Ahman 1994), over 330 kg of contaminated meat would have to be consumed to reach the maximum
intake level. Populations residing in the Marshall Islands were exposed to higher levels of ^{137}Cs than the
general population due to nuclear weapons testing conducted by the United States from 1946 to 1958.
Using radiological surveys from 1978 and 1985–1986, the AVDI of ^{137}Cs due to food ingestion was
estimated for seven age groups residing in the Rongelap Atoll (Robinson and Phillips 1989). The intakes
in pCi/day were as follows: 0–3 months of age, 424; 4–8 months of age, 556; 9 months to 1.4 years of
age 773; 1.5–3 years of age, 517; 4–11 years of age, 594; 12–17 years of age, 761; over 18 years of age,
1,085 (Robinson and Phillips 1989). This corresponds to an annual intake of 3.96×10^5 pCi/year for adults
over 18 years of age.

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 cesium is available. Where adequate information is not
available, ATSDR, in conjunction with the National Toxicology Program (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 cesium.

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 shown in Table 4-2, the relevant physical and chemical properties of cesium and its compounds are known (Burt 1993; Lide 1998). In general, cesium compounds are very water soluble and primarily exist in ionic form in aqueous environments. Cesium adsorbs strongly to soils and is not very mobile (Korobova et al. 1998; Paasikallio 1999; Ruse and Peart 2000; Takenaka et al. 1998). The radioactive decay modes of the two most important cesium isotopes, ^{134}Cs and ^{137}Cs, are also well known (ICRP 1983), and no additional physical or chemical data are required to predict the environmental fate of cesium.

Production, Import/Export, Use, Release, and Disposal.   Knowledge of a chemical's production volume is important because it may indicate possible 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 production volumes, import, and use of commercially significant cesium compounds are not available. No information was found for cesium in the TRI. According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit chemical release and off-site transfer information to the EPA. This database will be updated yearly and should provide a list of industrial production facilities and emissions. The United States is 100% import reliant for stable ^{133}Cs. No data are available on production volume, consumption, or import/export volumes of cesium. Although there is no information regarding the countries shipping cesium or cesium compounds to the United States, it is believed that Canada is the major source of cesium (USGS 1999). Data regarding the U.S. production (if any) of cesium and its compounds as well as import/export statistics would be useful. The amount of ^{134}Cs and ^{137}Cs released in airborne and liquid effluents from nuclear power plants in the United States is known (USNRC 1993).

Environmental Fate.   Information is available to permit the assessment of the environmental fate and transport of cesium in air (Hirose et al. 1993; Rybacek et al. 1994), water (Burt 1993; WHO 1983), and sediment and soil (Korobova et al. 1998; Paasikallio 1999; Ruse and Peart 2000; Takenaka et al. 1998). Cesium compounds are water soluble, and only one oxidation state (+1) is observed under environmental conditions (Burt 1993). Cesium released to the atmosphere may be carried long distances before being deposited onto soil and water surfaces by wet and dry deposition (Hirose et al. 1993). Most of the cesium released to water will adsorb to suspended solids in the water column and ultimately be deposited in the sediment core. In soil surfaces, cesium is strongly adsorbed in the upper layers and generally has very low mobility (Korobova et al. 1998; Paasikallio 1999; Ruse and Peart 2000; Takenaka et al. 1998). This is particularly true for soils with high potassium content or soils rich in clay. The radioactive decay modes of the two most important cesium isotopes, [134]Cs and ^{137}Cs, have been described in Chapter 4 (ICRP 1983).

Bioavailability from Environmental Media.   The bioavailability of cesium in environmental media is well understood and no data needs are identified at this time. For the most part, cesium is adsorbed strongly in the surface of most soils and is not readily bioavailable (Paasikallio 1999). In a study in which 102 healthy volunteers were fed [134]Cs-contaminated soil pellets, only about 1% of the original intake was absorbed, and on average, about 60% of the original amount was excreted within 48 hours (LeRoy et al. 1966). In peaty or podzolic soils with low clay content, cesium is reversibly adsorbed to the organic fraction and is expected to be in bioavailable form (Sanchez et al. 1999). Cesium uptake in vascular plants has been demonstrated (Djingova et al. 1995; Willey and Martin 1995). Increasing the clay or potassium content results in lower uptake by plants (Shenber and Johanson 1992). It has also been shown that fish residing in waters with high concentrations of humic material and potassium, such as oceans, have lower whole-body cesium concentrations than fish in freshwater where the concentration of dissolved potassium is lower for the same cesium concentration in the water (WHO 1983).

Food Chain Bioaccumulation.   Cesium bioaccumulates in both aquatic and terrestrial food chains (Rowan et al. 1998; WHO 1983). Mean BCFs for ^{137}Cs of 146, 124, and 63 were reported for fish, brown macroalgae, and molluscs, respectively (Fisher et al. 1999). Cesium accumulates in aquatic organisms from both food sources and from cesium dissolved in the water column or adsorbed to suspended solids and sediments. The lichen-caribou-human food chain is an important route of human exposure and has been well studied (Allaye-Chan et al. 1990; WHO 1983). No additional data are required to assess the potential for human exposure to cesium through food chain bioaccumulation.

6. POTENTIAL FOR HUMAN EXPOSURE

Exposure Levels in Environmental Media.   Stable cesium and radioactive cesium have been detected in air (Ajdacic and Martic 1990; Chiavarini et al. 1994; Dasch and Wolff 1989; Rybacek et al. 1994; Todorovic et al. 1999), water (Asubiojo et al. 1997; DOE 1992; Fisher et al. 1999; Prister et al. 1990; Strezov et al. 1999; WHO 1983), soil/sediment (Burt 1993; DOE 1992, 1998a; Ruse and Peart 2000; WHO 1983), and food (Ahman et al. 1994; Bibak et al. 1998; Mondon and Walters 1990). Due to the large surface area of lichens and moss, they can collect a great deal of atmospheric nuclear fallout and often have high concentrations of ^{134}Cs and ^{137}Cs (Papastefanou et al. 1992; Penttila et al. 1993). Grazing animals such as reindeer and caribou that feed on large quantities of lichens or moss may potentially ingest large amounts of radioactive cesium, and this cesium may be transferred to humans who consume these animals as a meat source (Allaye-Chan et al. 1990; WHO 1983). Continued monitoring data on ^{134}Cs and ^{137}Cs in environmental media are needed to increase knowledge of human exposure to ^{134}Cs and ^{137}Cs.

Exposure Levels in Humans.   Monitoring data are available for the levels of ^{134}Cs and ^{137}Cs in human populations (Glowiak et al. 1977b; Lokobauer et al. 1988; Rabitsch et al. 1991; Rahola and Suomela 1998; Ropolo and Cesana 1997). Most of the current data are for areas of eastern Europe and Russia. Limited data are available regarding the levels of ^{133}Cs in humans (Hewitt 1988). Stable cesium was detected in the urine of U.S. residents at geometric mean concentrations of about 4–5 μg/L depending upon age and gender (CDC 2003). More information regarding the background concentration of ^{133}Cs in human populations would be useful, but given the ubiquitous distribution of cesium at low levels in the environment, background levels are unlikely to approach levels that would reflect cesium toxicity. Since the levels of radioactivity change over time, continued monitoring data on the levels of ^{134}Cs and ^{137}Cs in humans are needed to evaluate potential risks to human health at current exposure levels.

Exposures of Children.   Children, like adults, are not expected to be over exposed to ^{133}Cs. Existing data show that children are exposed to ^{134}Cs and ^{137}Cs after major releases. These exposures are primarily related to dietary intake and the intake of contaminated milk (Lokobauer et al. 1988). Dietary consumption patterns suggest that the weight-adjusted intake of radioactive cesium for children may be greater than for adults following a nuclear release (Lokobauer et al. 1988). In a study of females from northern Sweden, ^{137}Cs was detected in breast milk from 8 out of 12 mothers at concentrations of 7.3–178.4 pCi/kg (Johansson et al. 1998). The infants of these mothers who were being breast-fed had whole-body levels of ^{137}Cs in the range of 45.9–675.7 pCi/kg (Johansson et al. 1998). Additional data regarding the transfer of ^{134}Cs and ^{137}Cs from breast milk and contaminated pasteurized milk samples to children, as

well as body burden studies on children, would be useful in assessing the potential risk that these radionuclides would pose following a major release.

Child health data needs relating to susceptibility are discussed in Section 3.13.2, Identification of Data Needs: Children's Susceptibility.

Exposure Registries.   Cesium is currently one of the chemicals for which a subregistry has been established in the National Exposure Registry. 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 cesium exposure.

6.8.2   Ongoing Studies

The database of federal research programs in progress (FEDRIP 2002) indicates several current projects that may fill some existing data gaps and add to the current database of knowledge. Studies are in progress to identify methods for more effective removal of radioactive cesium from contaminated soils and wastes. Researchers at the Universitaet des Saarlandes, Saarbruecken (Germany) are seeking cost-effective methods to remove gamma-emitting radionuclides, such as ^{137}Cs and ^{134}Cs, from radioactive waste commonly produced at nuclear power plants. Researchers at the Oak Ridge National Laboratory, Environmental Sciences Division are studying the environmental fate and behavior of selected radionuclides in terrestrial and aquatic environments. Dr. Michael Blaylock of Phytotech Incorporated is investigating the use of metal-accumulating plant species to extract and remove the radionuclides cesium and strontium from contaminated soils ("phytoextraction"). Rufus L. Chaney from the Beltsville Agricultural Research Center in Beltsville, Maryland is researching the feasibility of developing plants, soil, and plant management practices that can cost-effectively phytoextract radionuclides and heavy metals from contaminated soils. Dr. Heit from the U.S. DOE is conducting measurements regarding the fallout of nuclear debris from both atmospheric weapons tests and accidental atmospheric releases to determine the mechanisms of transport and deposition and to verify and correct fallout models. Work is being performed by Dr. Kinkead at the Los Alamos National Laboratory regarding the separation of cesium and strontium from high-level radioactive waste. Dr. Leon Kochian from the Agricultural Research Center in Ithaca, New York is investigating the bioaccumulation of ^{137}Cs in plants grown in contaminated soils.