6. POTENTIAL FOR HUMAN EXPOSURE

6.1 OVERVIEW

The presence of radon at any site can be a consequence of its natural occurrence in the environment plus any releases from anthropogenic hazardous waste.

The results of the 1992 EPA National Residential Radon Survey (EPA 1992b) estimated that 1 in 15 homes had an elevated radon level (i.e., a level at or above the EPA action level of 4 pCi/L). At the time, an estimated 5.8 million homes had an elevated radon level. The source of radon in homes is from naturally occurring (geologic) sources. For more information, refer to EPA's A Citizen's Guide to Radon (EPA 2009a).

^{222}Rn is a naturally occurring radioactive noble gas that is part of the ^{238}U decay chain, and is the daughter of ^{226}Ra. Similarly, ^{219}Rn and ^{220}Rn are in the ^{235}U and ^{232}Th decay chains and immediate daughters of ^{223}Ra and ^{224}Ra. ^{218}Rn is in the ^{238}U decay chain and the immediate daughter of ^{218}At. As radium decays, radon is formed and is released into small air or water-containing pores between soil and rock particles. If this occurs within radon's diffusion length of the soil surface, the radon may be released to ambient air (EPA 2003). Similarly, radon may migrate into groundwater. If this groundwater reaches the surface, some of the radon gas will release into the ambient air, but small amounts remain dissolved in the water. By far, the major sources of radon are its formation in and release from soil and groundwater, with soil contributing the greater amount (EPA 2003; Planinić et al. 1994). Radon is also released from the near surface water of oceans, tailings from mines (particularly uranium, phosphate, silver, and tin mines), coal residues, the combustion of fossil fuels (coal, oil, and natural gas), and building products (concrete, drywall, and brick) (Ericson and Pham 2001; Nero 1987). Global radon emissions from soil are estimated to be 2,400 million Ci ^{222}Rn (8,880x10^{16} Bq), followed by release from groundwater (500 million Ci), oceans (34 million Ci), phosphate residues (3 million Ci), uranium mill tailings (2 million Ci), coal residues (0.02 million Ci), natural gas emissions (0.01 million Ci), coal combustion (0.009 million Ci), and human exhalation (1x10^{-5} million Ci) annually (Fishbein 1992). Monitoring data in this chapter are reported for ^{222}Rn unless otherwise specified. The two other naturally occurring radioactive isotopes of radon, ^{219}Rn and ^{220}Rn, are not discussed due to their short half-lives (3.96 and 55.6 seconds, respectively; see Figures 4-2 and 4-3) (DOE 2008).

The ultimate fate of radon is transformation through radioactive decay. Radon decays only by normal radioactive processes (i.e., an atom of radon emits an alpha particle resulting in an atom of polonium,

which itself undergoes radioactive decay to other radon daughters or progeny) (EPA 2003). There are no sinks for radon, since its radioactive half-life is so short (3.8 days) (O'Neil et al. 2006).

In soil, radium atoms decay to radon, which can be released from the soil mineral matrix and transported through the soil column, ultimately being released to air. Alpha recoil is the process by which radon, when it is formed by radium emitting an alpha particle, actually recoils in the opposite direction from the path of particle ejection. Alpha recoil is important because this process dislodges radon from the edge of the soil mineral matrix and allows it to enter pore space between the soil grains. After radon is released into the pore spaces, its ultimate release to ambient air is a function of the soil porosity, soil moisture content, and meteorological factors, such as precipitation, atmospheric pressure, and the temperature versus altitude profile. Once radon is released to ambient air, its dispersion is primarily determined by atmospheric stability, including vertical temperature gradients and effects of wind. Transport of radon in indoor air is almost entirely controlled by the ventilation flow path and rate. Generally, the indoor radon concentrations increase as ventilation rates decrease. These transport processes are discussed in more detail in Section 6.3.1.

In groundwater, radon moves by diffusion and, primarily, by the mechanical flow of the water. Radon solubility in water is relatively low and, with its short radioactive half-life of 3.825 days (O'Neil et al. 2006), much of it will decay before it can be released from groundwater. Groundwater supplies in the United States have been surveyed for radon levels. In larger aquifers, average radon concentrations were reported to be 240 pCi (8.8 Bq)/L of water, while in smaller aquifers and wells, average levels were considerably higher (780 pCi/L of water; 28.9 Bq/L) (Cothern et al. 1986). These differences in radon levels between large and small groundwater supplies are a reflection of the types of rock and soil, as well as their uranium concentrations, through which the groundwater flows (Agency for Toxic Substances and Disease Registry 2011). Granitic rock, which is associated with high radon levels, does support large aquifers, although small aquifers may be present (Field and Kross 1998). For public groundwater-derived water supplies, the average radon concentration is estimated at 540 pCi/L (20 Bq/L), although some wells have been found to have radon concentrations up to 400 times the average concentration (up to $1x10^7$ Bq/m^3; 270,000 pCi/L). Surface water tends to have the lowest radon concentrations (NAS 1999b). Additional detail on radon in water is provided in Section 6.4.2.

Radon levels in ambient air vary with the type of soil and underlying bedrock of the area. The average outdoor radon concentration in the United States is about 0.4 pCi/L (14.8 Bq/m^3) (NAS 1999b). Measurements in Iowa and Minnesota show higher levels, with average outdoor concentrations of 0.60–

0.82 pCi/L (22.2–30.3 Bq/m^3) (Steck et al. 1999). Indoor concentrations as high as 2,000 pCi/L (74,000 Bq/m^3) have been observed in certain locations in the United States (EPA 2008b). Based on the National Residential Radon Survey, EPA estimates that the average indoor radon level is 1.25 pCi/L (46.25 Bq/m^3) in the United States (EPA 2003; Marcinowski et al. 1994); however, several locations in the country have been documented where the average indoor air levels are several times greater than the national average (Field 2005; Steck et al. 1999). The 1992 National Residential Radon Survey indicated that radon levels above the EPA recommended action level of 4 pCi/L could be present in 1 in 15 homes. At the time of the survey (1990), it was estimated that about 5.8 million homes had a higher radon level. For more information, refer to EPA's A Citizen's Guide to Radon (EPA 2009a).

Measurements of radon in soil are expressed in terms of levels in soil-gas. However, these measurements do not directly relate to rates of radon released to the atmosphere. Factors that affect radon soil-gas levels include soil properties such as radium content, mineral composition, moisture content, density, and soil porosity. Radon concentrations in soil may also be affected by meteorological conditions on the surface, such as snow (Fujiyoshi et al. 2002).

The primary pathway for human exposure to radon is inhalation from soil gas intrusion to dwellings and buildings; however, indoor radon levels can also originate from water usage, outdoor air infiltration, and the presence of building materials containing radium (EPA 2003). The committed dose from radon and its progeny is estimated by complex mathematical models and simplified tables have been published by EPA as Federal Guidance Report No. 13 (EPA 1999a). Exposure, both occupational and environmental, will be discussed primarily in terms of radon or radon progeny levels in the air. However, some estimates of daily intake can be made. For example, using an average indoor air radon concentration of 1.25 pCi/L (EPA 2003; Marcinowski et al. 1994) and an assumed breathing rate of 20 m^3/day, the radon daily intake from indoor air is 25,000 pCi/day. Using an estimated outdoor concentration of 0.4 pCi/L (NAS 1999b) and the same inhalation rate, the radon daily intake from outdoor air is 8,000 pCi/day.

Radon releases from groundwater also contribute to exposure. The daily intake of radon originating from drinking water only is estimated at 100–600 pCi (3.7–22.2 Bq)/day both from ingestion of drinking water and inhalation of radon released from drinking water (Cothern et al. 1986).

The highest occupational exposures to radon typically result from employment in underground uranium and other hard rock mining, or in phosphate mining due to the high airborne levels of radon and its progeny (NIOSH 2006). For example, an abandoned uranium mine located in Hungary had an average

radon concentration of 410 kBq/m^3 (11,100 pCi/L) at a depth of 15–55 m below the surface (Somlai et al. 2006). Although persons engaged in uranium mining are believed to receive the greatest exposures, the number of persons employed in uranium mining has greatly decreased. Furthermore, continuous improvements in engineering controls have lessened radon exposure in underground mines (NIOSH 1987). Measurements of radon progeny in U.S. mines from 1976 to 1985 showed annual mean concentrations of 0.11–0.36 working level (WL). A working level is "any combination of short-lived radon progeny in 1 liter of air that will ultimately release $1.3x10^5$ million electron volts of alpha energy during decay to lead-210" (NIOSH 1987). However, levels in phosphate mines measured during the same period showed a larger range of mean levels (0.12–1.20 WL) (NIOSH 1987). In 2006, assessments of radon exposure during phosphate plant operations resulted in an estimated mean concentration of 0.003 WL, based on limited data (NIOSH 2006).

While certain professions pose a higher risk of occupational exposure to radon (employment at underground mines for instance), exposure to high concentrations can occur in any location with geologic radon sources (Field 1999). A list of common occupations that have the potential for high radon and progeny exposure was developed by Field (1999). These occupations include mine workers (uranium, hard rock, and vanadium mines) and employees of water treatment plants, and radioactively contaminated sites can include uranium mill sites and associated mill tailing piles, phosphate fertilizer plants, oil refineries, power plants, and natural gas and oil piping facilities. Locations that are not contaminated, but at which elevated natural radon levels exist, can include natural caverns, utility and subway tunnels, excavation sites, health mines and spas, and fish hatcheries (EPA 2003; Field 1999; Fisher et al. 1996). Higher exposures can occur to farmers, radon mitigation professionals, and scientists studying radon or other radionuclides, although exposure to local radon sources occurs to everyone present, and elevated exposures can occur in any occupation (Field 1999).

6.2 RELEASES TO THE ENVIRONMENT

Manufacturing and processing facilities are required to report Toxics Release Inventory (TRI) to the EPA if specific criteria are met (EPA 2005). The TRI requirements do not apply to radon.

6.2.1 Air

There is no information on releases of radon to the atmosphere from manufacturing and processing facilities because these releases are not required to be reported (EPA 1998).

Because of the extended half-lives of uranium and radium and their abundance in the earth's surface, radon is continually being formed in soil and released to air. This normal emission of radon from ^{226}Ra in soils is the largest single source of radon in the global atmosphere (NAS 1999b; NCRP 1984a; Planinić et al. 1994). Using an average soil emanation rate of 1,600 pCi/cm^2-year and an estimated global surface area of $1.5x10^{18}$ cm^2, Harley (1973) estimated soil emanation of radon to be on the order of $2.4x10^9$ Ci ($8.9x10^{19}$ Bq)/year. Some solubilized radon is removed from the soil by plants through evapotranspiration where it is subsequently released to the atmosphere by diffusion through the leaf (Kozak et al. 2003; Taskayev et al. 1986).

Radon levels in outdoor air are affected by the composition of the substrate in the region. A monitoring study of radon in outdoor air conducted at 50 sites with varying geological characteristics in the state of Nevada indicated that the median statewide concentration of radon was essentially that of the nationwide average level of 0.4 pCi/L (Price et al. 1994). However, concentrations as large as 1.4 pCi/L were observed and these high levels usually correlated with silica rich igneous rocks (rhyolite and granite). Groundwater radon concentrations are also affected by the type of substrate. According to a study of North Carolina groundwater from private wells, areas with soil comprised on sand, silt, sandstones, and shales tend to have lower groundwater radon concentrations (67–1,700 pCi/L [2.5–63 Bq/L]) than groundwater in areas with metamorphic and granitic rocks (21–59,000 pCi/L [0.8–2,200 Bq/L]) (Watson et al. 1993).

Groundwater that is in contact with radium-containing rock and soil will be a receptor of radon emanating from the surroundings. When the groundwater reaches the surface by natural or mechanical means, this radon will start to be released to air. Although most of the radon present in groundwater will decay before reaching the surface, groundwater is considered to be the second largest source of environmental radon and is estimated to contribute $5x10^8$ Ci ($1.85x10^{19}$ Bq)/year to the global atmosphere (Fishbein 1992; NCRP 1984a). Radon is also released from oceans, but only from the near surface water, and in amounts that are an order of magnitude less than that from groundwater. As radium in oceans is largely restricted to bottom sediments, most radon would decay before water could carry it to the surface. Radon emissions from oceans were estimated as $3.4x10^7$ Ci/year (Fishbein 1992).

Radon in indoor air may also originate from volatilization of radon gas from water supplies used within homes for drinking, bathing, cooking, etc. Approximately 1–5% of the radon in indoor air was estimated to originate from water (Lam et al. 1994). Radon can also be released from water during the aeration and backwashing portions of the water treatment process. In a study of the water treatment process, exposure

to ^{222}Rn was measured at 31 Iowa water treatment plants found to have the largest decrease in ^{222}Rn water concentrations between raw and finished water. Workers were estimated to be exposed to an average annual air concentration of 3.4 pCi/L (126 Bq/m^3) ranging from 0.4 to 133 pCi/L (15–4,921 Bq/m^3). Facilities with the highest ^{222}Rn air concentrations treated groundwater containing moderate ^{226}Ra concentrations using aeration and iron filters. The estimated worker exposures were below the OSHA limit of 4 WLM/year based on short exposure intervals, even though exposures were overestimated by assuming radon-progeny equilibrium (Fisher et al. 1996)

Tailings from uranium mines and residues from phosphate mines each contribute to global radon in the approximate amount of 2–3x10^6 Ci (7.4x10^{16}–1.11x10^{17} Bq)/year, or a combined total of approximately 5x10^6 Ci (1.85x10^{17} Bq)/year. An abandoned mine in Hungary, with a subsurface radon concentration of 410 kBq/m^3 (11,100 pCi/L), was thought to have a significant effect on the air concentration of radon in houses above the mine. Indoor air concentrations, which averaged 667 Bq/m^3 (18.0 pCi/L), were likely elevated due to gas concentration within fissures reaching from the mine to the surface (Somlai et al. 2006). Fishbein (1992) reported that 3x10^6 Ci of ^{222}Rn is emitted from phosphate residues and 2x10^6 Ci of ^{222}Rn originates from uranium mill tailings each year.

Coal residues and fossil fuel (coal, oil, and natural gas) combustion products each contribute to atmospheric radon levels to a minor extent (NCRP 1984a). The portion from coal residues, such as fly ash, is very small. As natural gas retrieved from an area with concentrations of radium may contain high levels of radon, discharge via a combustion stream from a natural gas incinerator power plant may also have high radon levels. Emissions from one plant were measured as having an average concentration of 370 pCi/L (13,700 Bq/m^3). Radon is a noble gas, so it is not feasible to scrub it from any combustion stream. As of 2001, federal and State of California regulations did not control radioactive emissions such as these, which are considered to be “natural” emissions. Liquefied natural gas products from these sites may contain radon and progeny (Ericson and Pham 2001). Fishbein (1992) reported that coal residue and natural gas emissions release 20,000 and 10,000 Ci of ^{222}Rn each year, respectively, while coal combustion results in 900 Ci of ^{222}Rn production annually.

6.2.2 Water

There is no information on releases of radon to the water from manufacturing and processing facilities because these releases are not required to be reported (EPA 1998).

The amount of radon released to groundwater is a function of the chemical concentration of ^{226}Ra in the surrounding soil or rock and in the water itself (Hess et al. 1985). Radon can dissolve in groundwater following radioactive decay of the radium. High radon concentrations are associated with groundwater running over granitic rock or through alluvial soils originating from granite (Hess et al. 1985; Lam et al. 1994). The physical characteristics of the rock matrix are also important since it is believed that much of the radon released diffuses along microcrystalline imperfections in the rock matrix (Hess et al. 1985). Radon can also enter surface water through decay of radium.

6.2.3 Soil

There is no information on releases of radon to the soil from manufacturing and processing facilities because these releases are not required to be reported (EPA 1998).

As stated in Section 6.2.1, soil is the primary source of radon (NCRP 1984a; Planinić et al. 1994). As such, radon is not released to soil but is the result of radioactive decay of ^{226}Rn within the soil. Hopke (1987) states that normal soil-gas radon measurements are in the range of 270–675 pCi/L of air (10,000–25,000 Bq/m^3). However, levels exceeding 10,000 pCi/L of air (370,000 Bq/m^3) have been documented.

6.3 ENVIRONMENTAL FATE

6.3.1 Transport and Partitioning

The transport of radon from subsurface soil to air is a complex process that is dependent upon characteristics of the soil and meteorological conditions.

Emanation is the process by which radon is transported from the edge of a solid soil matrix to a gas or liquid pore space between the soil grains (Michel 1987). The mechanism by which this process occurs is primarily through alpha recoil. When a ^{226}Ra atom decays, it emits either a 4.6 or 4.8 MeV alpha particle, which results in the formation of a radon atom. The alpha particle takes a virtually straight line path in one direction, heavily ionizing the matrix in one direction and temporarily weakening the local mineral structure. At the same time, the radon atom experiences a 4.6 or 4.8 MeV equal, yet opposite reaction push, called a recoil, that physically moves the atom away from its original location. This recoil aids in moving a radon atom near the surface of a grain to a soil pore. The rate of emanation is typically slower in very dry soils since alpha recoil may also result in moving the recoiled atoms into an adjacent wall of another soil particle rather than an open pore space. On the other hand, if there is a small amount of water

in the pore space, the kinetic energy of the recoiling atom can be dissipated and radon atoms can be slowed sufficiently before becoming embedded into an adjacent soil particle. In a model developed to calculate radon emanation in soil, Sasaki et al. (2004) estimated that the alpha recoil range for radon was 0.02–0.07 µm in common minerals, 0.1 µm in water, and 63 µm in air. Once held within the pore space, radon may be transported by diffusion and convection to the surface where it is ultimately released to air.

The actual release of radon from the pore space or soil-gas to ambient air is called exhalation, while its release from water is called evaporation. The rates of these processes are functions of many variables including the concentration of radon in the soil-gas or water, soil porosity and moisture, meteorological factors (such as temperature and precipitation), and variations in atmospheric pressure (NAS 1999b; WHO 1983). Soil moisture has an important but varying effect on radon release to the air. While lower levels of soil moisture greatly increase emanation by preventing recoil atoms from embedding into adjacent walls of soil particles as described above, saturated soil conditions in which the pores are filled with water tend to slow the rate of diffusion to the surface since the diffusion coefficient of radon is about 3 orders of magnitude lower in water as compared to air (Markkanen and Arvela 1992; Michel 1987; WHO 1983). The influence of moisture and temperature on the radon exhalation rate in concrete, alum shale, and alum shale bearing soil was studied in laboratory experiments (Stranden et al. 1984). The results indicated that for each material, increasing the rate of moisture up to a certain point increased the radon exhalation rate from the material due to enhanced emanation. For concrete samples, the maximum exhalation rate occurred at a moisture content of 4.5–5.5%, for the alum shale, the maximum rate occurred at 10–15%, and for the soil samples, the maximum exhalation rate occurred at 20–30% moisture content (Stranden et al. 1984). As the moisture content increased beyond these levels, a dramatic decrease in the exhalation rate was observed. The authors concluded that when the pores were completely filled with water, the reduced rate of diffusion significantly attenuated the exhalation rate of radon from the material. If the porosity of the samples is high as in the case of the soil, more water can be absorbed by the sample before the pores are filled and the maximum rate of radon exhalation will occur at a higher moisture content than for low porosity materials.

Vertical temperature gradients in the atmosphere can create slight vacuum conditions that pull radon from the soil, or temperature inversions that inhibit this movement. Therefore, meteorological events may both enhance and inhibit transport of radon from the soil into other media. For instance, radon may be released from the soil surface into water from melting snow (Fujiyoshi et al. 2002). Alternatively, winter conditions may cause radon-containing soil-gas to become trapped in frozen soil, thus decreasing transmission of radon to the atmosphere (Bunzl et al. 1998).

Diurnal and seasonal changes affect the behavior of radon at the interface between soil and ambient air by impacting temperature and atmospheric mixing (NAS 1999b; UNSCEAR 2000). Once radon reaches a height of approximately 1 meter above the soil surface, its dispersion is predominantly determined by atmospheric stability (Cohen 1979). This stability is a function of vertical temperature gradient, direction and force of the wind, and turbulence. Temperature inversions in the early morning act to produce a stable atmosphere which keeps radon in the soil or near the ground or water surface. Solar radiation breaks up the inversion, leading to upward dispersion of radon which reverses with radiant cooling in late afternoon (Gesell 1983; NAS 1999b; UNSCEAR 2000). In general, radon levels in air typically decrease exponentially with altitude (Cohen 1979). In a study by Chandrashekara et al. (2006), outdoor radon concentrations at 1 meter above the ground were found to increase during the night, peak in the very early morning, and decrease during the day. In the United States, radon concentrations typically reach their maximum in the summer to early winter, whereas from late winter to spring, concentrations are usually at a minimum as a result of meteorological changes and soil moisture conditions (NAS 1999b).

Sources of indoor radon include entry of amounts released beneath the structure, entry in utilities such as water and natural gas, and release from building materials. Normally, the greatest contribution is that from radon released from soil or rock (Nero 1987; Planinić et al. 1994). Entry occurs primarily by bulk flow of soil-gas driven by small pressure differences between the lower and upper parts of the house interior and the outdoors. The pressure differences are primarily due to differences in indoor/outdoor temperature and the effects of wind (Nero 1987).

In cases where uranium or other metal mine or mill tailings are used for construction purposes, the primary source of indoor radon can be from these materials (Agency for Toxic Substances and Disease Registry 2006). Mill tailings are a rather uniform sand that may be superior to local supplies in quality and price. They have been used for under slab foundations, for concrete and mortar mix (used in laying foundations, block, brick, and stone work), and even as a supplement for vegetable gardens. Radon buildup in such homes, along with direct gamma emissions from radium and radon progeny, contribute to elevated radiation exposure.

Transport of radon in indoor air is primarily a function of the outflow ventilation rate of the enclosure. Most residential heating and air conditioning systems operate in a total recirculation mode, which doesn’t contribute to a ventilation rate. Under most conditions, the indoor radon concentration increases in direct proportion to the decrease in ventilation rates (WHO 1983). However, in some indoor radon studies,

radon concentrations showed greater variability than could be accounted for by ventilation rates. This was said to suggest that the strength of the radon source was the main cause of the wide range in observed indoor radon levels (Nero 1987). Behavior of radon in enclosed areas has also been extensively studied and predicted by modeling (Bowring 1992; Eichholz 1987; Kitto 2003).

Transport is primarily a function of the fraction of attachment of radon daughters to dust and dirt particles in the air, the concentration and size of the particles, and the rate of deposition. A major complication of modeling both radon and radon daughter transport indoors is that the outflow ventilation rate acts both to increase flow of radon into the structure and to remove radon and radon daughters from the structure through cracks and openings (Nero 1987). Air circulation rate also acts on the movement of air indoors causing variations in radon concentrations from room to room, as well as within a room.

Mechanisms for transport of radon in groundwater are complex. Just as transport in air is primarily governed by air flow patterns, the transport of radon in groundwater is accomplished by diffusion and, primarily, by the mechanical flow patterns of groundwater (Watson et al. 1993). As previously stated, the diffusion coefficient of radon in water is sufficiently low so that diffusion is only important for movement in very small and poorly ventilated spaces (such as pore spaces). The solubility of radon in water is relatively low (230 cm^3/L of water at 20 °C) and, due to radon's relatively short half-life, much of it will have decayed to polonium and other non-volatile progeny before the groundwater reaches the surface. However, that remaining in solution can be released to ambient air once it is encountered. In areas where groundwater has high levels of radon, release from groundwater may significantly affect ambient air levels.

6.3.2 Transformation and Degradation

6.3.2.1 Air

Regardless of the surrounding media, radon is a noble gas that transforms only by radioactive decay. There are no sinks for radon, and it is estimated that only negligible amounts escape to the stratosphere (Harley 1973). Therefore, the transformation of ^{222}Rn proceeds by alpha-emission with a half-life of 3.8235 days (NNDC 2012b). The half-lives of its first four progeny are much shorter, ranging from 164.3 µsec for ^{214}Po to 26.8 minutes for ^{214}Pb. The half-lives and progeny for ^{219}Rn, ^{220}Rn, and ^{222}Rn (as well as for all known radionuclides) are internationally maintained by DOE (NNDC 2012a) and are shown in Figures 4-1 through 4-3. NIST has developed and provides precise radon emanation rate standards in encapsulated solution form (currently, SRMs 4971, 4972, 4973, and 4974) for use in

calibrating radon monitors. Since ^{222}Rn standards are required for home radon testing, NIST has worked to transfer the U.S. national standards to secondary calibration laboratories (Kotrappa et al. 2005; NIST 2010).

6.3.2.2 Water

Radon undergoes natural radioactive decay in water by the mechanisms described in Chapter 4.

6.3.2.3 Sediment and Soil

Radon undergoes natural radioactive decay in soil by the mechanisms described in Chapter 4.

6.3.2.4 Other Media

Though radon is inert, it can react with highly electronegative elements, such as oxygen, fluorine, and chlorine, to form relatively stable compounds (Hwang et al. 2005; O'Neil et al. 2006). For example, radon reacts with fluorine to form radon fluoride, which has a fairly low volatility (Chernick et al. 1962).

6.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

Reliable evaluation of the potential for human exposure to radon depends in part on the reliability of supporting analytical data from environmental samples and biological specimens. Concentrations of radon in unpolluted atmospheres and in pristine surface waters are typically within the limits of current analytical methods. In reviewing data on radon 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 available. The analytical methods available for monitoring radon in a variety of environmental media are detailed in Chapter 7.

6.4.1 Air

Outdoor radon levels vary with geographic location and their proximity to radon sources in rocks and soil, water bodies, mines or mill tailings, and fossil-fuel combustion facilities (NAS 1999b). Gesell (1983) provided a compilation of data on radon levels in outdoor air. Measurements were taken over the continental United States, Hawaii, and Alaska. The highest concentrations were found in the Colorado Plateau, which is a region containing high levels of uranium as well as mines and uranium tailings. Measurements in this region ranged from 0.5 to 0.75 pCi/L of air (18.5–30 Bq/m^3). Average values from

the continental United States ranged from 0.12 to 0.3 pCi/L of air (4.4–11 Bq/m^3). More recent estimates based on an analysis of the available data of radon concentrations outdoors and on the transfer from water to air approximate the average outdoor air concentration over the entire United States as approximately 0.4 pCi/L (14.8 Bq/m^3) (NAS 1999b).

Price et al. (1994) reported the statewide median outdoor air concentration in Nevada to be 0.4 pCi/L (15 Bq/m^3), with a range of 0.07–1.40 pCi/L (2.6–52 Bq/m^3) for 50 sites. The ranges correlated to various concentrations of radon in soil as well as uranium and progeny in rocks. In Iowa and Minnesota, Steck et al. (1999) reported average outdoor radon concentrations of 0.82 pCi/L (30 Bq/m^3) and 0.60 pCi/L (22 Bq/m^3), respectively. Values in Iowa ranged from 0.2 to 1.5 pCi/L (7–55 Bq/m^3), while those in Minnesota ranged from 0.1 to 1.5 pCi/L (4–55 Bq/m^3).

Radon concentrations in air decrease with height from the soil surface (NAS 1999b). Several investigators have measured radon levels in the troposphere. Machta and Lucas (1962) measured 0.007 pCi/L of air (0.26 Bq/m^3) at 25,000 feet. Comparable measurements have been taken over Alaska and the southwestern United States (Harley 1973). Radon concentrations measured at a few centimeters above the ground surface may be a factor of 10 higher than measurements from 1 meter above the surface, although this factor would vary with atmospheric conditions (UNSCEAR 2000). The changes in radon concentration with height are thought to be the result of atmospheric conditions (mixing and turbulence) (NAS 1999b).

Numerous studies have been conducted to measure the radon concentrations of indoor air. Nero et al. (1986) reanalyzed up to 38 small data sets, of which 22 were considered unbiased. Biased data were those collected from areas where high radon concentrations were expected. On the basis of the unbiased data, the geometric mean of indoor radon levels was reported to be approximately 0.9 pCi/L of air (33 Bq/m^3). The arithmetic mean concentration was 1.5 pCi/L of air (56 Bq/m^3). Distribution studies of household levels indicated that from 1 to 3% of single-family houses may exceed 8 pCi/L of air (296 Bq/m^3). In this study, many of the measurements were made in main-floor living rooms or average living areas (Nero et al. 1986). On average the relative air concentrations of radon in residential dwellings are 1.8, 1.0, 0.9, and 0.5 pCi/L (66.6, 37, 33.3, and 18.5 Bq/m^3) for the basement, first, second, and third floors, respectively (Planinić et al. 1994), indicating that radon concentrations decrease with distance from the earth's surface. The National Residential Radon Survey conducted in 1989 and 1990 (published in 1992) determined that the indoor average concentration of radon for U.S. homes was approximately 1.25 pCi/L (46.3 Bq/m^3) (Marcinowski et al. 1994). Approximately 6% of homes studied

(5.8 million homes in 1990) had radon levels exceeding the EPA's recommended action level of 4 pCi/L (148 Bq/m^3) (Marcinowski et al. 1994).

A screening assessment conducted by the EPA of 55,000 homes located in 38 different states indicated that six counties in the Three Mile Island vicinity of Pennsylvania (Cumberland, Dauphin, Lancaster, Lebanon, Perry, and York) had the highest regional average indoor air levels of radon (17.8 pCi/L) (Field 2005). The author suggested that these high radon levels are the main source of radiation exposure to residents in this area and have not often been accounted for in epidemiological studies of residents in this area. Homes built in contact with bedrock may have a higher likelihood of elevated radon concentrations in indoor air. Brookins (1991) reported high indoor radon levels in residential dwellings of Albuquerque, New Mexico. These values correspond to high soil radon levels in the area, although they may have also been affected by the type of building materials used in the homes. Four of five adobe buildings showed radon levels >4 pCi/L (ranging from 2.0 to 10.7 pCi/L), while smaller percentages of homes utilizing other construction methods had elevated levels.

In an EPA assisted survey of indoor radon concentrations within 30 states, concentrations were found to vary widely between states. Additionally, houses with livable basements had higher radon concentrations than houses without basements. The mean concentration for those with basements ranged from 1.8 pCi/L (67 Bq/m^3) in Arizona and California to 9.4 pCi/L (348 Bq/m^3) in Iowa. Those without basements had mean concentrations ranging from 0.5 pCi/L (19 Bq/m^3) in Louisiana to 5.5 pCi/L (204 Bq/m^3) in Iowa (White et al. 1992).

Indoor radon levels were measured in homes located in the Reading Prong area of Pennsylvania. This area has an unusual abundance of homes with high radon concentrations that is presumed to be from geologically produced emanation of radon. Indoor levels of radon in this area ranged from 4–20 pCi/L (150–740 Bq/m^3) in 29% of the homes to >80 pCi/L (3,000 Bq/m^3) in 1% of the homes (Fleischer 1986). During a hot spot survey, indoor residential radon levels, also in the Reading Prong area, ranged from 0.2 to 360 pCi/L (Lewis 1996).

6.4.2 Water

In a nationwide survey by the EPA, almost 2,500 public drinking water supplies were sampled (nonrandom) with most of these serving greater than 1,000 people (Cothern et al. 1986). Average concentrations for U.S. groundwater were estimated to be 240 pCi/L of water (8.8 Bq/L) for larger

systems (>1,000 persons served) and 780 pCi/L of water (28.9 Bq/L) for smaller systems. The nationwide average for all groundwater samples tested in this study was 351 pCi/L (13 Bq/L). The highest levels reported were in smaller groundwater systems in Maine that averaged 10,000 pCi/L (370 Bq/L); lowest average levels were found in larger systems in Tennessee with levels of 24 pCi/L (0.9 Bq/L). Small, private groundwater systems appear to have higher radon concentrations than larger systems (Swistock et al. 1993; Watson et al. 1993). The average radon concentration in groundwater-derived public water supplies is approximately 540 pCi/L (20 Bq/L), although some public water supplies have been found to have radon concentrations up to $1x10^7$ Bq/m^3 (270,000 pCi/L) (NAS 1999b). Longtin (1988, 1990) has compiled the results of a comprehensive monitoring study (1984–1986) regarding the levels of radon, radium, and uranium in public drinking water supplies in the United States. The results indicated that over 72% of the sites sampled had radon concentrations greater than the minimum reporting limit of 100 pCi/L (3.7 Bq/L), and a maximum concentration of 25,700 pCi/L (951 Bq/L) was observed. The USGS conducted a comprehensive groundwater monitoring study (1992–2003) of aquifers across the United States for the presence of radon and various trace elements (USGS 2011). The median concentration of radon (n=3,877) was 430 pCi/L (15.8 Bq/L), with a maximum level of 220,000 pCi/L (8,140 Bq/L).

The relationship between radon concentrations in groundwater and system size (concentrations tend to increase with decreasing system size) was previously reported by Hess et al. (1985). This correlation may reflect a relationship between system size and aquifer composition. Those rock types that are associated with high radon levels (granitic rock) do not form aquifers large enough to support large systems. However, smaller systems may tap into such aquifers. Additionally, radon concentrations tend to decrease as the well depth increases, which may be attributed to the substrate composition at the various depths (Field and Kross 1998).

Crystalline aquifers of igneous and metamorphic rocks generally have higher radon levels than other aquifer types. Aquifers comprised of granites or alluvial soils derived from granite consistently show the highest levels (Lam et al. 1994; Michel 1987), though sandstone and feldspar substrates are also correlated to high radon levels (Lam et al. 1994). Average radon levels in water from granite aquifers are usually ≥2,703 pCi/L of water (100 Bq/L) (Michel 1987). This is indicated in the data of Cothern et al. (1986) which report the following trends in groundwater radon levels: in New England and the Piedmont and Appalachian Mountain Provinces, where igneous and metamorphic rocks form the aquifers, concentrations are in the range of 1,000–10,000 pCi/L of water (37–370 Bq/L); in the sandstone and sand aquifers that extend from the Appalachian Mountains west to the Plains, concentrations are generally

<1,000 pCi/L of water (37 Bq/L). NAS (1999b) also reported high radon concentrations in public water supplies for New England, the Appalachian states, and the Rocky Mountain states, as well as areas of the Southwest and Great Plains. A granitic substrate in the San Joaquin Valley of California contributes to high radon concentration in groundwater. The groundwater of several California counties contains levels of radon as high as 1,000–10,000 pCi/L (Lam et al. 1994).

A study of groundwater from 48 Pennsylvania counties indicated a median radon concentration of 1,100 pCi/L for all samples, with a maximum concentration of 141,270 pCi/L. The highest concentrations were present in samples obtained from Southeastern Pennsylvania, which includes geologic formations typical of high radon emission (Swistock et al. 1993). In North Carolina, the arithmetic mean radon concentration tested in groundwater supplies of 400 homes was 1,800 pCi/L (67 Bq/L) (Watson et al. 1993).

It has been reported that the radon concentration in surface waters is usually <4,000 Bq/m^3 (108 pCi/L) NAS (1999b).

6.4.3 Sediment and Soil

Because radon is a gas, its occurrence in soil is most appropriately referred to as its occurrence in "soil-gas," which is the gas or water-filled space between individual particles of soil. Factors that affect radon soil-gas levels include radium content and distribution, soil porosity, moisture, and density. However, soil as a source of radon is seldom characterized by radon levels in soil-gas, but is usually characterized directly by emanation measurements or indirectly by measurements of members of the ^{238}U series (NRC 1981). Radon content is not a direct function of the radium concentration of the soil, but radium concentration is an important indicator of the potential for radon production in soils and bedrock. However, Michel (1987) stated that average radium content cannot be used to estimate radon soil-gas levels, primarily due to differences in soil porosity. Similarly, Fujiyoshi et al. (2002) found that radium content may not control radon concentration in soil. In the study, radium concentrations were fairly consistent across various sites though the radon concentrations varied.

Despite such caveats, theoretical rates of radon formation in soil have been estimated as demonstrated by the following (Nevissi and Bodansky 1987):

> Consider a cube which is 1 meter in each dimension. Using rounded numbers, if the average density of the soil is 2.0 grams per cubic-centimeter and the average radium-226

concentration is 1.0 pCi/g (0.037 Bq/g), the cube will contain 2 million grams of soil and $2x10^{-6}$ Ci ($7.4x10^{4}$ Bq) of radium-226. This corresponds to the production of $7.4x10^{4}$ radon atoms per cubic-meter per second and the escape of 7,400 atoms per square-meter per second, in rough correspondence to the average measured value. In alternative units, the figure of 0.5 pCi per square-meter per second corresponds to the emission of 16 Ci of radon per square-kilometer per year.

For a discussion of ^{238}U and ^{226}Ra levels in soil, see the ATSDR Toxicological Profiles for uranium and radium (Agency for Toxic Substances and Disease Registry 1999a, 2011).

Brookins (1991) reported the average concentration of radon in soil-gas in the United States is approximately 100 pCi/L. However, this value does not compare well with two soil-gas measurements for U.S. locations found in the literature: one from Spokane, Washington, with soil-gas radon levels of 189–1,000 pCi/L (7,000–37,000 Bq/m^3) in soils formed from coarse glacial outwash deposits with 2.3 ppm uranium, and the other from Reading Prong, New Jersey, with soil-gas radon levels of 1,081–27,027 pCi/L of air (40,000–1,000,000 Bq/m^3) (Michel 1987). Hopke (1987) states that normal soil-gas radon measurements are in the range of 270–675 pCi/L (10,000–25,000 Bq/m^3).

Radon levels in soil-gas can fluctuate greatly, both temporally and spatially (Bunzl et al. 1998). A Bavarian study found that the concentration of radon in soil-gas of a high gravel content soil was higher at a depth of 0.5 m than at 1.0 m during the winter months, whereas in the summer, concentrations at the 1.0-m depth were higher. Bunzl et al. (1998) reasoned that high levels exhibited during the winter months were most likely the result of frozen soil conditions, whereby transmission of radon to the atmosphere is decreased and thus, levels in soil-gas are increased. The annual mean concentration at a depth of 0.5 m was observed to be 17.1 kBq/m^3 (462 pCi/L) while the mean level at a depth of 1.0 m was 15.2 kBq/m^3 (411 pCi/L) (Bunzl et al. 1998). At a depth of 38 cm, radon levels were found to range from 40 to 890 pCi/L in Albuquerque, New Mexico. The average summer value was 360 pCi/L, while the average winter levels were 200 pCi/L (Brookins 1991).

6.4.4 Other Environmental Media

Limited information exists to indicate that plants absorb both ^{226}Ra and ^{222}Rn from the soil layer and that these compounds are translocated to above ground plant parts (Taskayev et al. 1986). However, there is little information on the quantitative contribution of this process to exposure from ingestion of plant crops or of emanation rates from these plants. A measurement of the emission rates of radon from field corn was located in the literature. ^{222}Rn flux from leaves was reported to be $2.47x10^{-4}$ pCi

($9.15x10^{-6}$ Bq)/cm^2/second. This rate was 1.8 times greater than the exhalation rate from local soil (Pearson 1967). Solubilized radon can be removed from the soil by plants through evapotranspiration, where it is subsequently released to the atmosphere by diffusion through the leaf. Kozak et al. (2003) designed a flow and transport model to describe the transport or radon and radium through soil and vegetation.

6.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

In the following section, exposure to radon is discussed in terms of environmental levels rather than in terms of actual or estimated dose. The estimation of whole body or target tissue dose of radionuclides is extremely complex and must be accomplished by mathematical models for the specific radionuclide. Although such models are available to estimate whole body and target tissue dose for radon, discussion of these lies outside the scope of this document. For a discussion of these models, the reader is referred to NCRP (1984a) or NAS (1999a).

The general population is exposed to radon by inhalation, both outdoors and indoors, as well as by ingestion. Radon concentrations in outdoor air often correspond to soil gas levels (Price et al. 1994), although concentrations vary widely with geographical location, depending on factors such as the radium content, soil porosity, and moisture content. Comparing data from multiple studies, NAS (1999b) reports that the mean radon concentrations range from 1 to 63 Bq/m^3 (0.027–1.7 pCi/L) with the highest values reported in Iowa and Maine, with an overall average radon concentration of 0.32 pCi/L (12 Bq/L). Measurements in Iowa and Minnesota show average outdoor concentrations of 0.60–0.82 pCi/L (Steck et al. 1999). The average outdoor air concentration of radon over the entire United States is approximately 0.4 pCi/L (NAS 1999b). Due to the gaseous nature of radon, radon levels will decrease with increasing height from the soil surface; however, Price et al. (1994) reported that radon concentrations in Nevada obtained at heights of 0.5, 1.0, and 2.0 m from the surface were not statistically different from each other. This indicates that adults and children sitting or standing in the same location are exposed to similar concentrations.

Average radon levels indoors are found to be higher than ambient outdoor levels (Steck et al. 1999). When the general population encounters elevated concentrations of radon, it generally is while indoors, such as at home, school, or work where concentrations exceed the EPA-recommended action level of 4 pCi/L (CDC 1999). The National Residential Radon Survey conducted in 1989 and 1990 (published in 1992) determined that the indoor average annual concentration for U.S. homes was approximately

1.25 pCi/L (EPA 2003; Marcinowski et al. 1994). Approximately 6% of homes studied (5.8 million homes in 1990) had radon levels exceeding the EPA's action level of 4 pCi/L (Marcinowski et al. 1994). Two large indoor monitoring efforts in the United States reported arithmetic mean levels ranging from 1.5 to 4.2 pCi/L of air (55–157 Bq/m^3) (Alter and Oswald 1987; Nero et al. 1986). The data from Alter and Oswald (1987) are limited in that the dwellings do not represent a random sample and individual measurement values were reported rather than average concentrations from a residence.

The composition and physical properties of concrete, such as porosity, can affect the rate by which radon moves through an intact concrete slab and enters a home. Renken and Rosenberg (1995) estimated that a typical basement with a 1,500 ft^2 (140 m^2) concrete slab would have approximately 7.1 Bq/hour of radon diffusing through the concrete slab. Decreasing the porosity, permeability, and diffusion coefficient of the concrete mix can result in less radon gas diffusing through the slab and into the home.

Although the primary source of indoor radon is from soil, release of radon from water may contribute to indoor levels (Fishbein 1992; Lam et al. 1994). Nazaroff et al. (1987) performed an analysis that combined information on water use, efficiency of radon release from water, house volumes, and ventilation rates to determine the impact on indoor radon levels. Their analysis estimated that use of groundwater contributes an average of 2% to the mean indoor radon concentration in houses. Lam et al. (1994) concluded that groundwater may contribute 1–5% of indoor radon. As with levels in other media, levels of radon in groundwater vary greatly. In areas with high groundwater levels, the relative contribution to indoor radon levels will increase accordingly. Cothern et al. (1986) report a daily intake of radon originating from drinking water of 100–600 pCi (3.7–22.2 Bq)/day, assuming that consumption was 2 L/day of groundwater. Additionally, small groundwater systems appear to have higher radon concentrations than larger systems (Swistock et al. 1993).

The contribution of building materials to indoor radon (other than homes where metal mine or mill tailings have been used in construction) is estimated to be low in comparison with amounts which originate from soil and rock. In general, among common building materials, concrete and gypsum board release more radon than other materials.

The type of concrete used in a house slab can affect the rate at which radon diffuses from the ground through a basement slab and into the home. Renken and Rosenberg (1995) assessed porosity, permeability, and diffusion constants through three mix types. Diffusion constants in increasing order were 4.96×10^{-4} $cm^2 second^{-1}$ for a typical basement slab concrete mix, 9.09×10^{-4} $cm^2 second^{-1}$ for concrete

with an increased water:cement ratio, and $1.43x10^{-3}$ $cm^2second^{-1}$ for concrete with substituted fly ash. The respective porosities for these slabs were 0.12, 0.17, and 0.20. It was concluded that controlling the porosity of a concrete slab can reduce the rate of radon transmission into a house.

Active soil depressurization (ASD) was assessed for its effectiveness in mitigating radon in a home with basement concentrations averaging 7,580 Bq/m^3 (205 pCi/L). The system reduced levels to 520 Bq/m^3 (14 pCi/L). After a more powerful fan was installed to increase vacuum, radon levels in the basement unexpectedly increased to 1,070 Bq/m^3 (29 pCi/L). Upon reversal of the fan direction to produce an active soil pressurization system (ASP), the large fan reduced levels to 63 Bq/m^3 (1.7 pCi/L). Reinstallation of the small fan into the ASP system further reduced the radon level to 44 Bq/m^3 (1.2 pCi/L). The indications are that an overly forceful ASD vacuum can break the ground seal, reducing its effectiveness, and that ASP might be more effective than ASD in some cases (Kearney and Mason 2011).

Persons who are occupationally exposed to radon typically are those employed in mining and milling, primarily underground mining of uranium and hard rock (NIOSH 1987), but which also include silver, tin, bertrandite and beryl ores, and other mines (Kaczynski 2011; Lubin et al. 1994). Exposure to radon in underground mines has been shown by numerous studies to be a high risk factor for developing lung cancer (EPA 2003), particularly for miners in China, the Czech Republic, the United States, and Canada (Lubin et al. 1994). Exposures in above-ground mines and in mills are typically lower.

NIOSH reports that in 2005, 22,838 workers were employed in underground metal and nonmetal mines in the United States, with 29,705 workers employed at all underground mines (including metal, nonmetal, coal, and stone mines) (NIOSH 2008a). In 2005, 263 metal mines and 739 nonmetal mines were reported (NIOSH 2008b). The number of underground uranium mines has decreased from 300 in 1980 to 16 in 1984 (NIOSH 1987) to 17 in 1992 (EPA 1995), although the number may have increased to <100 in 2003 (IAEA 2004). The number of employees in underground uranium mines has decreased from 9,000 in 1979 to 448 in 1986 (NIOSH 1987), although figures were not available for later years. Measurements of radon progeny concentrations in these mines from 1976 to 1985 showed annual geometric mean concentrations in uranium mines of 0.11–0.36 WL (equivalent to 22–72 pCi/L of air [800–2,664 Bq/m^3] assuming an equilibrium factor of 0.5), with 95^{th} percentile levels ranging up to 2.73 WL (546 pCi/L of air; 20,202 Bq/m^3). Annual geometric mean levels in phosphate mines for the same period were 0.12–1.20 WL (24–240 pCi/L of air [888–8,880 Bq/m^3]) with 95^{th} percentile levels as high as 1.69 WL (338 pCi/L of air; 12,506 Bq/m^3). Measurements in uranium/vanadium mines showed annual geometric

mean concentrations similar to those in uranium mines. However, 95th percentile levels ranged up to 4.80 WL (960 pCi/L of air [$3.6x10^4$ Bq/m^3]), which was the highest annual concentration reported among the different types of mines (NIOSH 1987). Estimates of annual cumulative radon progeny exposures indicated that of the 1,405 underground uranium miners working in 1984, 28% had exposures >1 WL (200 pCi/L of air; 7,400 Bq/m^3). As uranium is a minor impurity in bertrandite and beryl ores, radon may be present above ambient levels where these ores are processed, such as at a beryllium extraction facility located in Delta, Utah (Kaczynski 2011).

Radon exposure in underground mines has been vastly reduced by installation of improved engineering controls. In New Mexico mines, the median annual exposure in 1967 of 5.4 WLM was reduced to 0.5 WLM by 1980 due to these improvements (Eichholz 1987). For 1982, Samet et al. (1986) reported a mean WLM of 0.7. A WLM expresses both intensity and duration of exposure (see Chapter 3 for further discussion).

MSHA regulates safety practices and worker protection in the mining industry. OSHA has established air monitoring requirements for underground mines and exposure limits for mine workers. These involve monitoring mine exhaust air for radon daughters, with values >0.1 WL for areas where uranium is mined (or between 0.1 and 0.3 WL for areas where uranium is not mined) requiring periodic monitoring of air representative of the workers' breathing zones (MSHA 2011c). Workers are not to be exposed to concentrations exceeding 1.0 WL in any active mine area (MSHA 2011b). In cases where accepted engineering control measures have not been implemented or when work conditions require, higher-level exposure is permitted under an appropriate respiratory protection program (MSHA 2001d). The goal is to ensure that no underground mine worker receives >4 WLM in any calendar year (MSHA 2011a).

Occupational exposure to radon can extend beyond mining. Water-plant operators may be exposed to high levels of radon gas created during the water treatment process. This occurs when radon emanates from water to air during the aeration process or when filter material to strip out uranium or radium is removed for disposal as radioactive waste. The geometric annual mean air concentration of radon in 31 water plants was 3.4 pCi/L (126 Bq/m^3), with a maximum value of 133 pCi/L (4,921 Bq/m^3) (Fisher et al. 1996). A high exposure risk is also present for employees at radioactive contaminated sites, nuclear waste repositories, natural caverns, phosphate fertilizer plants, oil refineries, utility and subway tunnels, excavators, power plants, natural gas and oil piping facilities, health mines and spas, fish hatcheries, and hospitals (EPA 2003; Field 1999; Fisher et al. 1996). Higher exposure risks are also present for farmers,

radon mitigation professionals, and scientists, although exposure to local radon sources can occur in any occupation (Field 1999).

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 1993).

Exposure levels at schools were utilized to provide an estimate of radon levels to which children may be exposed during the school day. However, limited U.S. data were available to address radon exposure of children.

The EPA recommends that all schools test for radon and mitigate areas with elevated concentrations. EPA's 1990 National School Radon Survey obtained radon measurements from 927 randomly selected schools across the United States. Based on these measurements, it is estimated that approximately 15,000 U.S. schools have at least one room with a potential for long-term elevation of radon levels. Radon is often unevenly distributed within a building. Overall, short-term radon concentrations in roughly 2.7% of all ground contact schoolrooms were >4 pCi/L, indicating 73,000 schoolrooms with a potential radon problem (EPA 1993c).

Additionally, higher respiration rates of children may influence the extent of radon and radon progeny inhaled. MacDonald and Laverock (1998) studied the exposure levels of soil-dwelling mammals in a radon-rich environment, concluding that larger mammals with higher lung capacities were least affected by radon. Most affected were smaller mammals with higher respiration rates. Using this logic, small

children with high respiration rates, as compared to adults, may receive relatively higher radiation doses from inhaled radon and radon progeny.

Kendall and Smith (2005) examined the doses of radon and its decay products inhaled or ingested by 1-year-old infants and 10-year-old children in the United Kingdom. The largest internal doses were found to be associated with the organ of intake (the respiratory tract and stomach). Dose coefficients (or the dose per unit intake factors) were found to be higher for children than for adults, although the overall annual doses were fairly consistent between children and adults (likely due to the smaller amount of air and water consumed by children).

6.7 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Populations with potentially high exposures include those occupationally exposed. Those who use excavation equipment or are employed at underground mines (uranium, hard rock, and vanadium), water treatment plants, radioactively contaminated sites, natural caverns, phosphate fertilizer plants, oil refineries, utility and subway tunnels, fossil fueled power plants, natural gas and oil piping facilities, health mines and spas, and fish hatcheries have the potential to be more highly exposed to radon (EPA 2003; Field 1999; Fisher et al. 1996). Higher exposures are also possible for farmers, radon mitigation professionals, and scientists (Field 1999).

High radon exposure can occur in any location with geologic radon sources (see http://www.epa.gov/radon/zonemap.html) (EPA 2011a; Field 1999). High outdoor air radon concentrations were reported in Iowa, Main, and Minnesota NAS (1999b). NAS (1999b) also reported high radon concentrations in public water supplies for New England, the Appalachian states, and the Rocky Mountain states, as well as areas of the Southwest and Great Plains. Though the average radon concentration in groundwater-derived public water supplies is approximately 540 pCi/L (20 Bq/L), some public water supplies have been found to have radon concentrations up to $1x10^7$ Bq/m^3 (270,000 pCi/L) (NAS 1999b).

Communities that are very near uranium or phosphate mill tailing piles may have increased environmental radon levels. In addition, in some areas, mill tailings have been used as fill dirt, garden soil, sub-base for concrete slabs, and sand mix for brick mortar in home construction (for example, in Monticello, Utah) (Agency for Toxic Substances and Disease Registry 1997). Persons in these communities could be exposed to levels of radon exceeding typical indoor background levels.

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 radon 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 radon.

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. Information is available on the physical and chemical properties of radon, and parameters that influence the behavior of radon in the environment have been determined. Therefore, no data needs are identified concerning physical and chemical properties of radon.

Production, Import/Export, Use, Release, and Disposal. According to the Emergency Planning and Community Right-to-Know Act of 1986, 42 U.S.C. Section 11023, industries are required to submit substance release and off-site transfer information to the EPA. The TRI, which contains this information for 2006, became available in March of 2008. This database is updated yearly and should provide a list of industrial production facilities and emissions.

The production of radon occurs directly from a radium source either in the environment or in a laboratory environment. The disposal of gaseous radioactive effluents has been documented. Increased radon concentrations have been detected in waste generated by uranium and phosphate mining; therefore, these sites should be monitored on a continual basis. Although there are regulations for disposal of radionuclides in general, there are none that specifically address disposal of materials due to their radon

content. If such regulations were promulgated, they would be developed by states since the Federal government has no authority in this area.

Environmental Fate. Information is available on the environmental fate of radon in air and water and on the transport of radon in environmental media. Factors that affect the partitioning of radon from soil or water to air have been identified. Movement of radon into and within homes and the influence of meteorological conditions and other parameters on this movement should continue to be investigated. Transformation of radon has been adequately characterized. There is limited information on the uptake and release of radon by plants. Additional research of this phenomenon is needed in order to determine the relative contribution plants provide to atmospheric levels. Exposure from smoking tobacco should be explored.

Bioavailability from Environmental Media. Radon and radon progeny are known to be released from air and water and information is available, which characterizes the relative contribution of various media to levels of radon in air and water.

Food Chain Bioaccumulation. Since radon is a noble gas, it will not bioaccumulate. However, bioaccumulation has been reported for radon progeny such as ^{210}Pb in cephalopods (Khan and Wesley 2011) and ^{210}Po in marine birds (Skwarzec and Fabisiak 2007), mushrooms (Skwarzec and Jakusik 2003), cephalopods (Khan and Wesley 2011), and coastal sand dune wild legumes (Bhat et al. 2005).

Exposure from Environmental Media. Reliable monitoring data for the levels of radon in contaminated media at hazardous waste sites might be helpful, particularly if uranium mine tailings have been disposed of at these sites.

Information is available regarding the levels of radon in indoor air, outdoor air, and water. Continued comprehensive data on levels of radon in ambient air are needed in order to assess potential human exposure. The measurement of indoor and ambient radon levels are not mandated, and EPA has found that most homeowners do not choose to spend the money to have these measurements made.

Exposure Levels in Humans. EPA maintains information on those states and jurisdictions that have enacted Radon-Resistant New Construction building codes (EPA 2011g). Large-scale monitoring of radon in public buildings (e.g., schools) was conducted in the 1990s. Limited information for the United States in general is available on remediation activities conducted in response to those measurements and

the resulting radon levels, or on radon levels in building constructed since that time. Radon is a naturally occurring gas and is ubiquitous in the environment; therefore, humans are constantly exposed to some level of radon. The primary pathway for human exposure to radon is inhalation from soil gas intrusion to dwellings and buildings. Outdoor radon levels vary with geographic location and their proximity to radon sources in rocks and soil, water bodies, mines or mill tailings, and fossil-fuel combustion facilities. Since the half-life of radon is short, its measurement in biological samples, such as serum, urine, blood, etc., is not practical. Concentrations of radon progeny are measurable in urine, blood, bone, teeth, and hair, and these levels can be used to provide some indication of exposure; however, they are not direct measurements of levels of exposure. These estimates may be derived through use of mathematical models.

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

Exposures of Children. Limited information is available to address radon exposure of children, particularly within the United States. Some communities require testing of schools for radon and abatement if levels are ≥4 pCi/L (NJDEP 2004). Available data were not always in agreement, and thus, conclusions were difficult to assess. Studies are needed to better characterize exposure levels specific to children in the United States.

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 radon 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.

The Hanford Environmental Foundation in Richland, Washington, maintains a registry of United States uranium miners and millers. The data in the registry are derived from autopsy material and include exposure information. Since uranium decays to radon, this exposure registry on miners and millers may provide information on radon exposure. The NIOSH dose reconstruction and worker compensation programs should also be addressed.

6.8.2 Ongoing Studies

No ongoing studies were identified.