5. POTENTIAL FOR HUMAN EXPOSURE

5.1 OVERVIEW

Radon is a product of the natural radioactive decay of uranium, which occurs naturally in the earth's crust, to radium and then to radon. As radium decays, radon is formed and is released into small air or water-containing pores between soil and rock particles. If this occurs near the soil surface, the radon may be released to ambient air. Radon may also be released into groundwater. If this groundwater reaches the surface, most of the radon gas will quickly be released to ambient air, but small amounts may remain in the water, By far, the major source of radon is its formation in and release from soil and groundwater, with soil contributing the greater amount. Smaller amounts of radon are released from the near surface water of oceans, tailings from mines (particularly uranium and phosphate mines), coal residues and combustion products, natural gas, and building products, such as concrete and brick.

The ultimate fate of radon is transformation through radioactive decay. Radon decays only by normal radioactive processes, that is, an atom of radon emits an alpha particle resulting in an atom of polonium, which itself undergoes radioactive decay to other radon progeny. There are no sinks for radon; therefore, small amounts of radon are lost to the stratosphere.

In soil, radon is transported primarily by alpha recoil and mechanical flow of air and water in the soil. 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. After radon is released into the pore spaces, its ultimate release to ambient air a function of the soil porosity and meteorological factors, such as precipitation and atmospheric pressure. 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 rate in the enclosure. Generally, the indoor radon concentrations increase as ventilation rates decrease.

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 3.8 days, much of it will decay before it can be released from groundwater.

Radon levels in ambient air vary with the type of soil and underlying bedrock of the area. Available measurements indicate that the mean value for atmospheric radon in the contiguous United States is approximately 0.25 pCi radon-222/L of air (9 Bq/m^3). However, measurements of air from the Colorado Plateau show radon levels up to 0.75 pCi radon-222/L of air (30 Bq/m^3). Studies of indoor radon levels indicate an average concentration of from 1.5

to 4.2 pCi radon-222/L of air (55 to 157 Bq/m^3) (Alter and Oswald 1987; Nero et al 1986).

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) radon-222/L of water, while in smaller aquifers and wells average levels were considerably higher (780 pCi radon-222/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 type of rock which surrounds them.

Measurements of radon in soil are expressed in terms of levels in soilgas. However, these measurements do not directly relate to rates of radon released to the atmosphere. Factors which affect radon soil-gas levels include radium content, soil porosity, moisture content, and density. Technically, measurement of soil-gas is difficult and there are few studies which report such data.

Delivered dose of radon and its progeny can only be estimated by complex mathematical models. Therefore, exposure, both occupational and environmental, will be discussed, primarily in terms of radon levels in the air. However, some estimates of daily intake have been made. Daily intake of radon originating outdoors is estimated to be 970 pCi (36 Bq) radon-222/day (Cothern et al. 1986). Exposure from indoor radon is higher due to concentration of levels from lack of ventilation and other factors. Total daily intake of radon originating indoors is estimated as 8,100 pCi (300 Bq) radon-222/day, assuming a breathing rate of 20 m^3/day. However, daily intake is dependent on time spent in and outdoors and on breathing rate (Cothern et al. 1986).

Radon releases to the environment (primarily indoor levels) from groundwater also contribute to environmental exposures. The daily intake of radon originating from drinking water only is estimated at 100 to 600 pCi (3.7 to 22.2 Bq) radon-222/day both from ingestion of drinking water and inhalation of radon released from drinking water (Cothern et al. 1986). Radon releases from building materials contribute little to potential exposure.

Occupational exposure to radon results from employment in uranium and other hard rock mining, or in phosphate mining. Persons engaged in uranium mining are believed to receive the largest exposures, although the number of persons employed in uranium mining has steadily decreased in the past 9 years. Measurements of radon progeny in these mines from 1976 to 1985 showed annual mean concentrations of 0.11 to 0.36 WL (22 to 72 pCi radon-222/L of air; 800 to 2,664 Bq/m^3) (NIOSH 1987). However, levels in phosphate mines measured during the same period showed a larger range of mean levels (0.12 to 1.20 WL; 24 to 240 pCi radon-222/L of air; 888 to 8,880 Bq/m^3). Radon exposure in underground mines is continually being reduced due to improved engineering controls (NIOSH 1987).

5.2 RELEASES TO THE ENVIRONMENT

5.2.1 Air

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 emanation of radon from radium-226 in soils is the largest single source of radon in the global atmosphere (NCRP 1984a). Using average emanation rates from available measurements, Harley (1973) estimated soil emanation of radon to be on the order of $2x10^{9}$Ci ($7.4x10^{19}$ Bq) radon-222/year. This estimation is equivalent to 1,600 pCi (60 Bq/cm^2) radon/cm^2 soil/year (Harley 1973). The emanation rate at a particular location is highly variable and is affected by many factors, including barometric pressure, composition of soil, and soil moisture and temperature. Usually, less than 10% of radon in upper soil layers is released to the atmosphere (Vilenskiy 1969). Some radon is released by plants through evapotranspiration. However, the amount released has not been estimated (Taskayev et al. 1986).

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 man-made forces, this radon will be released to air. Although most of the radon present in groundwater will decay before reaching the surface, groundwater is still considered to be the second largest source of environmental radon and is estimated to contribute $5x10^{8}$ Ci ($1.85x10^{19}$ Bq) radon-222/year to the global atmosphere (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. Radium in oceans is largely restricted to the sediments where it cannot affect atmospheric levels of radon (Harley 1973).

Tailings from uranium mines and residues from phosphate mines both contribute to global radon in the approximate amount of 2 to 3 x 10^{6} Ci (7.4×10^{16} to 1.11×10^{17} Bq) radon-222/year. Although these sites are not numerous (in 1984 there were 50 sites containing uranium tailings), emanation rates to air may be substantial. It is estimated that 20% of the radon formec in tailings is released and that emanation rates can be as high as 1,000 pCi (37 Bq) radon/m^2/second (NCRP 1984a).

Coal residues and combustion products, as well as natural gas, each contribute to atmospheric radon levels to a minor extent (NCRP 1984a). Coal and natural gas, at the time of combustion, release radon to air. Coal residues, such as fly ash, contribute very small amounts to atmospheric radon.

Some building materials release very small amounts of radon. However, the major source of radon in single family dwellings is the soil directly under the building (NCRP 1984b).

According to the VIEW database, 14 NPL sites reportedly contain radon above background levels (VIEW 1989). The frequency of these sites within the United States can be seen in Figure 5-1. Quantification of the levels found is not available. However, the majority of the radon released would be to air.

5.2.2 Water

The amount of radon released to groundwater is a function of the chemical concentration of radium-226 in the surrounding soil or rock and in the water itself. High radon activity is associated with groundwater surrounded by granitic rock. The physical characteristics of the rock matrix are important also since it is believed that much of the radon released diffuses along microcrystalline imperfections in the rock matrix (Hess et al. 1985). Radon is rarely found in surface water due to the fact that it is rapidly released to the air when the water reaches surface levels (Michel 1987).

In a reanalysis of published data, Hess et al. (1985) reported a geometric population average of 187 pCi (6.9 Bq) radon-222/L of water in over 6,000 samples of groundwater supplies for public use. In contrast, samples of surface water supplies indicated that the average level of radon was 1 pCi (0.037 Bq) radon-222/L of water.

5.2.3 Soil

As stated in Section 5.2.1, soil is the primary source of radon. As such, radon is not released to soil but is the result of radioactive decay of radium-226 within the soil. The radon concentration in the soil is a function of the radium concentration, the soil moisture content, the soil particle size, and the rate of exchange of air with the atmosphere (Hopke 1987). Hopke (1987) states that normal soil-gas radon measurements are in the range of 270 to 675 pCi radon-222/L of air (10,000 to 25,000 Bq/m^3). However, levels exceeding 10,000 pCi radon-222/L of air (370,000 Bq/m^3) have been documented.

5.3 ENVIRONMENTAL FATE

5.3.1 Transport and Partitioning

Emanation is the process by which radon is transported from a solid to a gas or liquid medium. At the soil particle level, radon gas is transferred from soil particles into pore spaces (gas- or liquid-filled spaces between soil particles) primarily by alpha recoil. Alpha recoil occurs after radium decays by emitting an alpha particle. After the particle is ejected, the resulting radon atom actually recoils in the opposite direction. Alpha recoil results in breaking of chemical bonds in the solid, physically moving the atom to a different position, and damaging the crystal structure. The radon atom may recoil to a position from which it will not be released (embedded in the same particle or in another particle) or may recoil into the pore space from

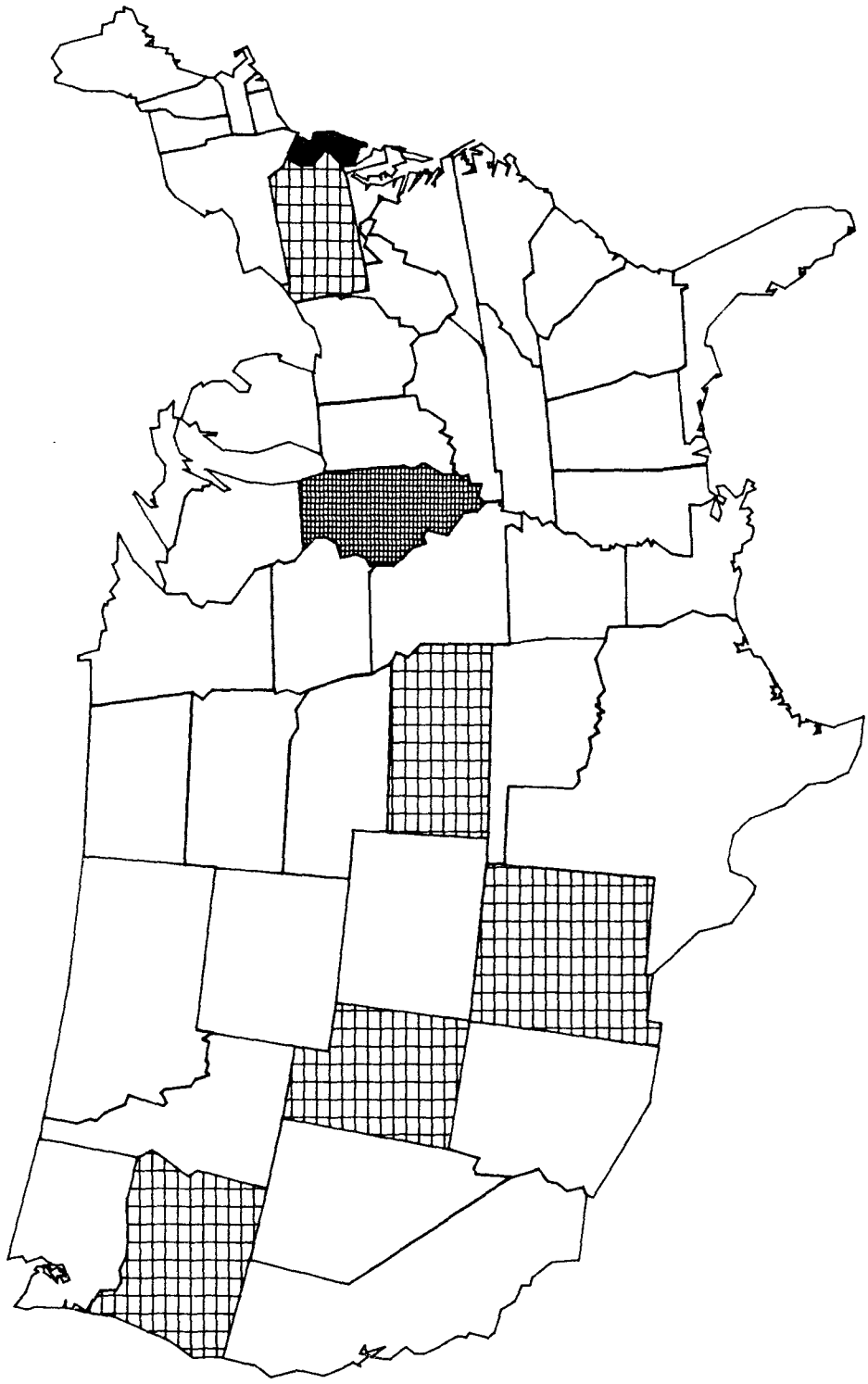

FIGURE 5-1. FREQUENCY OF SITES WITH RADON CONTAMINATION

which it may move by diffusion or convection toward the soil surface. If the pore space is filled with liquid, any radon atoms which recoil into it will travel slower than those that recoil into air-filled spaces (Michel 1987). Although alpha recoil is believed to be the major process of radon release from solids, diffusion from very small pores near the particle surfaces and along imperfections of the crystalline structure of the particle also occurs.

Once radon enters the pore space, it is transported by diffusion, convection, and flow of rain and groundwater. The diffusion constant for radon is approximately 10^{-2} cm^2 per second in air and 10^{-5} cm^2 per second in water (WHO 1983). These constants indicate that diffusion of radon is a relatively slow process and that its movement is, therefore, primarily accomplished by mechanical transport of air and water in the pore space.

The actual release of radon from the pore space or soil-gas to ambient air is called exhalation. The rate of this process is a function of many variables including the concentration of radon in the soil-gas, the soil porosity, and meteorological factors such as precipitation and variations in atmospheric pressure (WHO 1983).

Behavior of radon at the interface between soil and ambient air is not well understood. However, 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 concentrations near the ground. Solar radiation breaks up the inversion, leading to upward dispersion of radon which reverses with radiant cooling in late afternoon (Gesell 1983). In addition, general trends in air turbulence lead to maximum levels in air in the early autumn and early winter (when turbulence is generally less) and lower levels in air in the spring due to increased turbulence (Michel 1987). In the absence of these factors, radon levels in air decrease exponentially with altitude (Cohen 1979). This phenomenon has been studied by sampling and many models have been derived to fit the data (WHO 1983).

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. The greatest contribution is that from radon released from soil or rock (Nero 1987). Entry occurs primarily by bulk flow of soilgas driven by small pressure differences between the lower part 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).

Transport of radon in indoor air is primarily a function of the ventilation rate of the enclosure. 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 (Eichholz 1987; Jonassen 1975).

Transport of radon daughters indoors has also been extensively modeled. Transport is primarily a function of the rate of attachment of radon daughters to particles, 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 ventilation rate acts both to increase flow of radon into the structure and to remove radon and radon daughters from the structure (Nero 1987). Ventilation 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 much less complex than those for other media. In fact, transport of radon in groundwater is accomplished by diffusion and, primarily, by the mechanical flow of groundwater. As previously stated, the diffusion coefficient of radon in water is sufficiently low so that diffusion is only important for movement in very small spaces (such as pore spaces). The solubility of radon in water is relatively low (230 cm^3 radon-222/L of water at 20°C) and, due to radon's relatively short half-life, much of it will have decayed before the groundwater reaches the surface. However, that remaining in solution will be quickly 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.

5.3.2 Transformation and Degradation

5.3.2.1 Air

Regardless of the surrounding media, radon is transformed or degrades 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, degradation proceeds by alpha-emission to form polonium-218. As stated in Table 3-2, the half-life of radon is 3.82 days. The half-lives of the progeny are much shorter, ranging from approximately 0.0002 seconds for polonium-214 to 30 minutes for lead-214.

5.3.2.2 Water

See Section 5.3.2.1.

5.3.2.3 Soil

See Section 5.3.2.1.

5.4 LEVELS MONITORED OR ESTIMATED IN THE ENVIRONMENT

5.4.1 Air

The most comprehensive compilation of data on radon levels in outdoor air was reported by Gesell (1983). 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 radon-222/L of air (18.5 to 30 Bq/m^3). Average values from the continental United States ranged from 0.12 to 0.3 pCi radon-222/L of air (4.4 to 11 Bq/m^3). Based on these and other data, Michel (1987) states that the mean value for atmospheric radon in normal geological areas of the contiguous United States is approximately 0.25 pCi radon-222/L of air (9 Bq/m^3) with a range of 0.1 to 0.4 pCi radon-222/L of air (4 to 15 Bq/m^3).

Data reported by Fisenne (1987) indicate variability of radon levels with time. In continuous data (9 years of hourly measurements), both diurnal and seasonal patterns were observed. Diurnal variations showed an early morning peak and a drop in the afternoon. Seasonally, levels were highest in early autumn and lowest in early spring.

Radon concentrations in air decrease with height from the soil surface. Several investigators have measured radon levels in the troposphere. Machta and Lucas (1962) measured 0.007 pCi radon-222/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).

Although there are many studies which undertake to quantify radon in indoor air, the work of Nero et al. (1986) is the most comprehensive and the most often cited. This study 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 radon-222/L of air (33 Bq/m^3). These data implied an arithmetic average concentration of 1.5 pCi radon-222/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 radon-222/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).

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 to 20 pCi/L (150 to 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).

Other studies of indoor radon levels were summarized in NCRP (1987). The median levels ranged up to 18.9 pCi radon-222/L of air (700 Bq/m^3) in homes in Butte, Montana (Israeli 1985). A study by Cohen (1986) reported results from 453 indoor sites in 42 states and showed a mean of 1.62 pCi radon-222/L of air (60 Bq/m^3) with a median of 1.08 pCi radon-222/L of air (40 Bq/m^3).

5.4.2 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. Results of this survey were used to estimate the mean population-weighted radon levels in public groundwater systems by state (Cothern et al. 1986). Average concentrations for United States groundwater were estimated to be 240 pCi radon-222/L of water (8.8 Bq/L) for larger systems (>1,000 persons served), and for smaller systems 780 pCi radon-222/L of water (28.9 Bq/L). The nationwide average for all groundwater samples tested was 351 pCi radon-222/L of water (13 Bq/L). When surface water supplies were taken into consideration, due to the fact that their radon levels are essentially zero, the average radon concentration in all community water supplies was estimated to range from 54 to 270 pCi radon-222/L of water (2 to 10 Bq/L) (Michel 1987). The highest levels reported were in smaller groundwater systems in Maine which averaged 10,000 pCi radon-222/L of water (370 Bq/L); lowest average levels were found in larger systems in Tennessee with levels of 24 pCi radon-222/L (8.9 Bq/L).

This same relationship, i.e., radon concentrations in groundwater increasing with decreasing system size, was previously reported by Hess et al. (1985). This correlation is believed to 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.

Crystalline aquifers of igneous and metamorphic rocks generally have higher radon levels than other aquifer types with granites consistently showing the highest levels. Average radon levels in water from granite aquifers are usually 2,703 pCi radon-222/L of water (100 Bq/L) or greater (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 to 10,000 pCi radon-222/L of water (37 to 370 Bq/L); in the sandstone and sand aquifers which extend from the Appalachian Mountains west to the Plains, concentrations are generally less than 1,000 pCi radon-222/L of water (37 Bq/L).

5.4.3 Soil

Because radon is a gas, its occurrence in soil is most appropriately referred to as its occurrence in “soil-gas”, which is in the gas or waterfilled 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 uranium-238 series (National Research Council 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) states that average radium content cannot be used to estimate radon soil-gas levels, primarily due to differences in soil porosity.

Despite such caveats, theoretical rates of radon formation in soil have been estimated as demonstrated by the following:

> "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 cubiccentimeter and the average radium-226 concentration is 1.0 pCi/g (0.037 M/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." (Nevissi and Bodansky 1987).

For a discussion of uranium-238 and radium-226 levels in soil, see the ATSDR Toxicological Profiles for Uranium and Radium (ATSDR 1990a, 1990d).

Only two soil-gas measurements for United States locations were found in the literature: one from Spokane, Washington, with soil-gas radon from 189 to 1,000 pCi radon-222/L of air (7,000 to 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 from 1,081 to 27,027 pCi radon-222/L of air (40,000 to 1,000,000 Bq/m^3) (Michel 1987). Hopke (1987) states that normal soil-gas radon measurements are in the range of 270 to 675 pCi radon-222/L of air (10,000 to 25,000 Bq/m^3). It is reported that radon-222 levels increase with soil depth, reaching a probable maximum at about 800 cm below ground level (Jaki and Hess 1958).

5.4.4 Other Media

Limited information exists to indicate that plants absorb both radium-226 and radon-222 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. One measurement of emanation rates from field corn was located in the literature. Radon-222 release from leaves was reported to be 2.47×10^{-4} pCi ($9.15x10^{-6}$ Bq)/cm^2/sec. This emanation rate was 1.8 times greater than the emanation rate from local soil (Pearson 1967).

5.5 GENERAL POPULATION AND OCCUPATIONAL EXPOSURE

In the following section, exposure to radon is discussed in terms of environmental levels not 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 BEIR IV (1988).

The general population is exposed to radon by inhalation both outdoors and indoors. Outdoor levels, also referred to as ambient or background levels, are the result of radon emanating from soil. These levels vary widely with geographical location, depending on factors such as the radium content of soil and soil porosity and moisture content. However, a reasonable average for near ground level is suggested by Eichholz (1987) to be on the order of 0.150 pCi radon-222/L of air (5.55 Bq/m^3). Michel (1987) states that the mean value for atmospheric radon in the contiguous United States is approximately 0.24 pCi radon-222/L of air (8.88 Bq/m^3) with a range of 0.11 to 0.41 pCi radon-222/L of air (4.07 to 15.2 Bq/m^3). Cothern et al. (1986) report a daily intake of radon originating outdoors of approximately 1,000 pCi (36 Bq) radon-222/day based on data derived from the United Nations Scientific Committee on the Effects of Atomic Radiation (1982) and assuming an inhalation rate of 20 m^3/day of air containing 0.05 pCi/L (1.8 Bq/m^3) radon-222. Because of the gaseous nature of radon, radon levels will decrease with increasing height from the soil surface. Studies of this vertical gradient indicate that a child who is 0.5 m tall would be exposed to 16% more radon than an adult who is 1.5 m tall (Michel 1987).

In contrast to the average ambient levels of radon, which are usually quite low, levels indoors are found to be greater than ambient outdoor levels. This is due to enhancement and it is believed to be a function of the following: movement of radon from underlying soil and rock through the foundation of the building, release of radon from water and utility use, radon emanation from radium-containing structural materials, and rate of ventilation (NCRP 1984b). The contribution of each of these to the overall indoor radon level is difficult to assess, except qualitatively. It has been determined that elevated indoor radon levels are primarily due to radon emanation from underlying soil (Eichholz 1987). The actual indoor levels are greatly affected by other parameters such as composition of the foundation materials and the ventilation rate of the enclosed area. Two of the largest indoor monitoring efforts in the United States reported arithmetic mean levels ranging from 1.5 to 4.2 pCi radon-222/L of air (55 to 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. Cothern et al. (1986) report daily intake of radon originating indoors of 8,100 pCi (300 Bq) radon-222/day based on data derived from the

United Nations Scientific Committee on the Effects of Atomic Radiation (1982) and assuming indoor radon levels of 0.4 pCi radon-222/L of air (15 Bq/m^3).

Although the primary source of indoor radon is emanation from soil, release of radon from the water supply may contribute to indoor levels. Nazaroff et al. (1987) performed an analysis which 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. 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 to 600 pCi (3.7 to 22.2 Bq) radon-222/day assuming that consumption was 2 L/day of groundwater.

Radon ingestion from drinking water has been of less concern relative to the dose received from inhalation. Due to the short residence time in the stomach, ingested radon contributes a small dose to the stomach when compared to that delivered to the lungs from inhalation of radon released from water. Although the dose to the stomach is small, it is not significant (Eichholz 1987).

The contribution of building materials to indoor radon is estimated to be low in comparison with amounts which emanate directly from soil and rock. In general, among common building materials, concrete releases more radon than other materials. The potential of radon release from building materials is expressed by the radium content and the radon emanation rate, which is a function of pressure, temperature, porosity, and radon concentration. The radon emanation rate of concrete utilized in the United States is estimated to be in the range of 1.2 to 3.4×10^{-4} pCi (4.3 to 12.6×10^{-6} Bq) radon-222/kg/second (Michel 1987).

Persons who are occupationally exposed to radon are those employed in mining, primarily mining of uranium and hard rock (NIOSH 1987). NIOSH reports that in 1986, 22,499 workers were employed in metal and nonmetal mines in the United States. However, the number of underground uranium mines has steadily decreased from 300 in 1980 to 16 in 1984. In turn, the number of employees in underground uranium mines has decreased from 9,000 in 1979 (3,400 of whom worked underground 1,500 hours or more) to 448 in 1986 (NIOSH 1987). Measurements of radon progeny concentrations in these mines from 1976 to 1985 showed annual geometric mean concentrations in uranium mines of 0.11 to 0.36 WL (equivalent to 22 to 72 pCi radon-222/L of air [800 to 2,664 Bq/m^3] assuming an equilibrium factor of 0.5), with 95th percentile levels ranging up to 2.73 WL (546 pCi radon-222/L of air; 20,202 Bq/m^3). Annual geometric mean levels in phosphate mines for the same period were 0.12 to 1.20 WL (24 to 240 pCi radon-222/L of air [888 to 8,880 Bq/m^3]) with 95th percentile levels as high as 1.69 WL (338 pCi radon-222/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 radon-222/L of air [3.6×10^4 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 greater than 1.0 WL (200 pCi radon-222/L of air; 7,400 Bq/m^3).

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 this technique (Eichholz 1987).

Several researchers have attempted to correlate levels of lead-210 in bone with cumulative radon daughter exposure. Eisenbud et al. (1969) employed in vivo techniques to measure lead-210 in the skull of nonoccupationally exposed and occupationally exposed individuals. Exposure for miners was derived from mine records and compared to that estimated from a model. Their results showed that the amount of lead-210 deposited, regardless of temporal considerations, will be within a factor of two of that deposited if exposure is assumed to be uniform over time. In addition, they reported that a burden of 2,000 pCi (74 Bq) is equivalent to a calculated cumulative exposure of approximately 800 WLM.

Blanchard et al. (1969) reported a positive correlation between the log of lead-210 concentration in post-mortem derived bone and the log of estimated miners' cumulative exposure. However, more lead-210 was observed in bone than was predicted by the model utilized. Furthermore, a linear correlation was observed between lead-210 levels in blood and that in bone; however, for both of these analyses sample numbers were small (n=11 to 22). Another study (Clemente et al. 1984) has analyzed the correlation between lead-210 in human teeth and environmental radon levels in various countries. This analysis reported that for the incremental increase in lead-210 in teeth, a value of 3.24×10^{-3} pCi (1.2×10^{-4} Bq) radon-222/gm of tissue has been associated with a lifetime exposure to 1 WLM. All of these studies are limited by the difficulty in estimating exposure to individuals on the basis of mine levels and worker histories (often related by next of kin). Such estimates, although unavoidable, introduce considerable uncertainty into these analyses. In addition, lead-210 can be introduced in cigarette smoke, food, and ambient air, thus confounding results of studies (NCRP 1984b).

5.6 POPULATIONS WITH POTENTIALLY HIGH EXPOSURES

Populations with potentially high exposures include those occupationally exposed as previously described (see Section 5.5). In addition, certain populations are exposed to elevated environmental levels, such as those resulting from emanation from soil in the Reading Prong area of Pennsylvania (soil-gas of up to 27,000 pCi [1.0×10^3 Bq] radon-222/L soil-gas) and from release from groundwater in certain areas in Maine (levels up to 180,000 pCi [6.7×10^3 Bq] radon-222/L of water) (Hess et al. 1983). 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 for landfills and were subsequently developed (for example, Grand Junction, Colorado). Persons in these communities could be exposed to levels of radon exceeding normal background levels.

5.7 ADEQUACY OF THE DATABASE

Section 104(i)(5) of CERCLA 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 the 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 or eliminate the uncertainties of human health assessment. In the future, the identified data needs will be evaluated and prioritized, and a substance-specific research agenda will be proposed.

5.7.1 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, Use, Release, and Disposal. 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 radon contaminated materials. Further research on the disposal of radon attached to charcoal, which is used in radon monitoring indoors, would be beneficial.

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 which affect the partitioning of radon from soil or water to air have been identified. However, rates of flux from one media to another are rarely reported. The emanation rate of radon from soil is uncertain. Additional information on the behavior of radon at the soil-air interface, as well as soil-gas measurements, would facilitate a better understanding of the

emanation rate of radon from soil. Movement of radon into and within homes and the influence of meteorological conditions on this movement should be investigated. Study of radon movement would enhance understanding of potential indoor exposures. Transformation of radon has been adequately characterized. There is limited information on the uptake of radon by plants. Additional research of this phenomenon is needed in order to determine the effects of exposures which might be incurred from ingestion of food.

Bioavailability from Environmental Media. Radon and radon progeny are known to be absorbed from air and water and information is available which characterizes the relative contribution of various media to levels of radon in air and water. Further studies of bioavailability are not necessary at this time.

Food Chain Bioaccumulation. Information on bioaccumulation of radon and radon daughters in the food chain is not available. Therefore, the potential for bioconcentration in plants, aquatic organisms or animals, or for biomagnification in the food chain is unknown, However, due to the short half-life of radon, it would not tend to bioaccumulate. Studies of the bioaccumulation of radon in the food chain are not necessary at this time.

Exposure Levels in Environmental Media. Some information is available on exposure levels in environmental media, however, most of this information is from areas with higher than average levels of radon. Although levels in groundwater, primarily for public water supplies, have been more comprehensively reported than levels in ambient air, on-going monitoring efforts for both media are necessary for quantification of human exposure. Comprehensive data on levels of radon in ambient air are needed in order to assess potential human exposure.

Exposure Levels in Humans. There is a lack of comprehensive information associating radon and radon progeny levels monitored in the environment and exposure of the general population. Although levels of radon may be measured in exhaled air, the relationship of that amount exhaled to the exposure level can be estimated only by use of mathematical models. Concentrations of radon progeny are measurable in urine, blood, bone, teeth, and hair; however, these levels are not direct measurements of levels of exposure. These estimates also may be derived through use of mathematical models. Studies are needed to characterize the utility of these biomarkers of exposure.

Exposure Registries. No exposure registries for radon were located. This compound is not currently one of the compounds for which a subregistry has been established in the National Exposure Registry. The compound will be considered in the future when chemical selection is made for subregistries 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 the exposure to this compound. 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.

5.7.2 On-going Studies

S.D. Schery (New Mexico Institute of Mining and Technology) is studying the fundamental processes influencing release of radon isotopes from porous media and the physical properties of radon isotopes which affect their behavior in enclosed environments. The hypothesis that plant functions increase soil-radon flow to the atmosphere, thus measurably reducing the flow into subsurface areas is under investigation by F.W. Whicker (Colorado State University). A.B. Tanner (U.S. Department of Interior) is completing a qualitative study on the range and variability of diffusive and advective/convective transport of radon and its controlling factors at selected areas. Further, a study designed to provide information on the transport pathway of radon and radon progeny, their charge state, and the effect of clustering on decay products is being conducted by M.G. Payne (Oak Ridge National Laboratories). Computer models are also being developed in an attempt to simplify studies on radon transport within and from soils into the atmosphere and structures (P.C. Owczarski, Pacific Northwest Laboratories) and to unify theories of radon emanation and transport in the soil (K.K. Nielson, Rogers and Associates Engineering Corporation).

Investigations of factors which influence transport or mobility of radon and its progeny from rocks/soils to the environment or homes are underway by K.K. Turekian (Yale University), D. Thomas (University of Hawaii at Manoa), R.H. Socolow (Center for Energy and Environmental Studies, Princeton University), and C.S. Dudney (Oak Ridge National Laboratories), The study by Dudney included New Jersey and the Tennessee Valley areas with high background levels. The influence of season, heating fuel, tobacco smoking, and building characteristics on indoor air pollutant levels is being studied by R.H. Rainey (Office of Power, Tennessee Valley Authority). Research Triangle Institute (Research Triangle Park, North Carolina) is presently studying both modeling and measurement of radon in houses.

One aspect of radon mobility, in relation to groundwater, is being studied by O.S. Zepecza (U.S. Geological Survey). He is determining the factors which control radionuclide transport and fate in groundwater in the Newark basin and southern coastal plains of New Jersey, and the mechanism of release of radionuclides to groundwater or retention in aquifer solids.

G. Harbottle (Brookhaven National Laboratories) is studying the mobility and chemical behavior of radium in the soil and the processes involved in the emanation of radon. The dynamic behavior of radon and radon daughters will be studied in controlled laboratory environments (J.S. Johnson, Lawrence Livermore National Laboratory).