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I. Introduction

Radon is a natural radioactive gas that you cannot see, smell, or taste and that can only be detected with special equipment. It is produced by the radioactive decay of radium, which in turn is derived from the radioactive decay of uranium. Uranium is found in small quantities in all soils and rocks, although the amount varies from place to place. Radon decays to form radioactive particles that can enter the body by inhalation. Inhalation of the short-lived decay products of radon has been linked to an increase in the risk of developing cancers of the respiratory tract, especially of the lungs. Breathing radon in the indoor air of homes contributes to about 15,000 lung cancer deaths each year in the United States and 1100 in the UK (HPA, 2009). Only smoking causes more lung cancer deaths.

Geology is the most important factor controlling the source and distribution of radon (Appleton and Miles, 2010). Relatively high levels of radon emissions are associated with particular types of bedrock and unconsolidated deposits, for example some, but not all, granites, phosphatic rocks, and shales rich in organic materials. The release of radon from rocks and soils is controlled largely by the types of minerals in which uranium and radium occur. Once radon gas is released from minerals, its migration to the surface is controlled by the transmission characteristics of the bedrock and soil; the nature of the carrier fluids, including carbon dioxide gas and groundwater; meteorological factors such as barometric pressure, wind, relative humidity, and rainfall; and soil permeability, drainage, and moisture content. The most important factors controlling the migration and accumulation of radon in buildings include (i) characteristics of the bedrock and soils that affect fluid transport, including porosity and permeability; (ii) the construction of the building and its use which includes the level of ventilation and heating; and (iii) environmental factors such as temperature (increased heating in buildings during the colder months causes a chimney effect which draws soil gasses including radon into the property), plus wind speed and direction which can increase the chimney effect. Radon-containing soil air can be drawn into buildings through gaps and cracks in solid floors, walls and service pipes below construction level; through the voids in suspended

floors and crawl spaces, and via small cracks or pores in hollow-block walls or wall cavities. (see also Chapter 9, this volume).

Radon levels in outdoor air, indoor air, soil air, and groundwater can be very different. Radon released from rocks and soils is quickly diluted in the atmosphere. Concentrations in the open air are normally very low and probably do not present a hazard. Radon that enters poorly ventilated buildings, caves, mines, and tunnels can reach high concentrations in some circumstances. The construction method and the degree of ventilation can influence radon levels in buildings. A person's exposure to radon will also vary according to how particular buildings and spaces are used.

The radon potential of the ground may be assessed from a geologically based interpretation of indoor radon measurements in conjunction with permeability, uranium, soil gas radon, and ground and airborne gamma spectrometric data. The categorization of a group of rocks or unconsolidated deposits as having known or suspected high levels of radon emissions does not imply that there is any problem. That would depend on whether pathways, locations for accumulation, and protracted exposure occur. Whereas geological radon potential maps do not give a direct guide to the level of radon in specific buildings or cavities, there is, in general, a higher likelihood that problems may occur at specific sites within areas of potentially high radon emissions.

Radon potential maps have important applications, particularly in the control of radon through environmental health and building control legislation. They can be used (1) to assess whether radon protective measures may be required in new buildings, (2) for the cost-effective targeting of radon monitoring in existing dwellings and workplaces, (3) to provide a radon potential assessment for homebuyers and sellers, and (4) for exposure data for epidemiological studies of the links between radon and cancer. Whereas a geological radon potential map can indicate the relative radon hazard, it cannot predict the radon risk for an individual building. This can only be established by having the building tested.

Radon dissolved in groundwater migrates over long distances along fractures and caverns depending on the velocity of fluid flow. Radon is soluble in water and may thus be transported for distances of up to 5km in streams flowing underground in limestone. Radon remains in solution in the water until a gas phase is introduced (e.g., by turbulence or by pressure release). If emitted directly into the gas phase, as may happen above the water table, the presence of a carrier gas, such as carbon dioxide, would tend to induce migration of the radon. This appears to be the case in certain limestone formations, where underground caves and fissures enable the rapid transfer of the gas phase. Radon in water supplies can result in radiation exposure of people in two ways: by ingestion of the water or by release of the radon into the air during showering or bathing, allowing radon and its decay products to be inhaled. Radon in soil under homes is the biggest source of radon in indoor air, and it presents a greater risk of cancer than radon in drinking water.

This chapter explains how radon forms, the associated health risks, the kinds of rocks and soils it comes from, and how it moves through the ground and into buildings. It also explains how the radon potential of an area can be estimated. Greater detail on radon health risks, radon epidemiology, radon measurement methods, radon preventive measures in buildings, cost-effectiveness of radon control, radon risk communication and national radon programmes can be found in WHO (2009) and HPA (2009). (HPA, 2009; WHO, 2009).

II. Nature and Measurement

A. Radioactivity and Radiation

All matter, including the materials that constitute the Earth's crust, consists of atoms, which are usually combined in various chemical compounds. Each atom comprises a nucleus, made up of protons, neutrons, and electrons, which orbit around the nucleus. Nuclei identified by the name of the element and the number of protons and neutrons are referred to as nuclides. All nuclei of the same chemical element have the same number of protons, but they can have different numbers of neutrons, and these are then called isotopes of that element. Many atoms are unstable and will change quite naturally into atoms of another element accompanied by the emission of ionizing radiation. This process is called radioactivity and the change is called radioactive decay. Unstable atoms that

change through radioactive decay to form other nuclides are said to be radioactive and are referred to as radionuclides or radioisotopes. The rate of change or decay of an unstable radionuclide is indicated by its half-life, which is the period of time during which half the original number of atoms will have decayed.

The radiations most commonly emitted by radionuclides are alpha particles, beta particles, and gamma rays. The principal geological sources of radiation are *gamma radiation* from the ground and buildings and *radon gas*, which is derived mainly from uranium minerals in the ground. *Terrestrial gamma rays* originate chiefly from the radioactive decay of the natural potassium, uranium, and thorium, which are widely distributed in terrestrial materials including rocks, soils, and building materials extracted from the Earth.

There are three naturally occurring radon (Rn) isotopes: ^{219}Rn (actinon), ^{220}Rn (thoron), and ^{222}Rn , which is commonly called radon. ^{219}Rn (actinon) has a very short half-life of about 3 seconds and this, together with its occurrence in the decay chain of ^{235}U (which is only present as 0.7% of natural uranium), restricts its abundance in gases from most geological sources. Actinon does not escape to air in significant quantities. ^{222}Rn (radon) is the main radon isotope of concern to man. It occurs in the uranium-238 decay series (Table I), has a half-life of 3.82 days and provides about 50% of the total radiation dose to the average person. ^{222}Rn is produced by the radioactive decay of solid radium (^{226}Ra). ^{220}Rn (thoron) is produced in the thorium-232 decay series (Table II). ^{220}Rn has been recorded in houses, and about 4% of the average total radiation dose for a member of the UK population is from this source.

Most of the radon that is inhaled is exhaled again before it has time to decay and irradiate tissues in the respiratory tract. Radon (^{222}Rn), however, decays to form very small solid radioactive particles, including polonium-218, that become attached to natural aerosol and dust particles. These may remain suspended in the air or settle onto surfaces. When these particles are inhaled, they irradiate the lining of the bronchi in the lung with alpha particles and this may increase the risk of developing lung cancer.

B. Measurement of Radioactivity

There are a number of different ways to measure radioactivity. These include (1) the *radioactivity* of a radioactive material, such as radon gas; (2) the *dose* to living tissue, e.g., to the lungs from solid decay products of radon gas; and (3) the *exposure* caused by the presence of radioactivity. There are also environmental or safety thresholds of radioactivity such as *dose limit*, *action level*, and *reference level*, which are used in legislation and advice. The units of radioactivity and dose are summarized in Table III. In the United States, radioactivity is commonly measured in pico curies (pCi), named after the French physicist Marie Curie, who was a pioneer in the research on radioactive elements and their decay. In most other countries, and throughout this chapter, radioactivity is measured using the SI unit becquerel (Bq). One becquerel represents one atomic disintegration per second. The level of radioactivity in the air due to radon is measured in becquerels per cubic meter (Bq m^{-3}) of air. The average radon concentration in houses in Great Britain is 20Bq m^{-3} , that is, 20 radon atoms disintegrate every second in every cubic meter of air. The average in the United States is 46Bq m^{-3} . A 1000-square foot house with 46Bq m^{-3} of radon has nearly 2 million radon atoms decaying in it every minute.

Radon levels in outdoor air, indoor air, soil air, and groundwater can be very different. Radon in outdoor air is generally low ($4\text{--}8\text{Bq m}^{-3}$) but may be as high as 100Bq m^{-3} in some valleys when measured in the early morning. Radon in indoor air ranges from less than 20Bq m^{-3} to about $110,000\text{Bq m}^{-3}$ with a population-weighted world average of 40Bq m^{-3} and country averages ranging from 9 in Egypt, 20 in the UK, 22 in New Zealand, 44 in China, 46 in the United States, 108 in Sweden, 120 in Finland to 140Bq m^{-3} in the Czech Republic (UNSCEAR, 2009; HPA 2000). Radon concentrations in outdoor air in the UK are generally low, on average 4Bq m^{-3} whilst radon in indoor air in UK dwellings ranges from less than 10Bq m^{-3} to over $17\,000\text{Bq m}^{-3}$ (Rees et al., 2011). Radon in soil air (the air that occupies the pores in soil) ranges from less than 1 to more than 2500Bq L^{-1} ; most soils in the United States contain between 5 and 55Bq L^{-1} radon in soil air. The amount of radon dissolved in groundwater ranges from about 3 to nearly $80,000\text{Bq L}^{-1}$.

The *absorbed dose* is the energy absorbed by a unit mass of tissue whereas the *dose equivalent* takes account of the relative potential for damage to living tissue of the different types of radiation. The dose equivalent is the absorbed dose multiplied by a "quality factor," which is 1 for beta and gamma rays and 20 for alpha particles.

This is because alpha particles deposit their energy more densely. In addition, alpha particles transfer all their energy in short distances so that a relatively small volume of tissue receives a high dose of radiation. The commonly used unit for dose equivalent is the *sievert* (1Sv = 1000 millisieverts; mSv).

The dose equivalent indicates the potential risk of harm to particular tissues by different radiations, irrespective of their type or energy. *Risk weighting factors* are an approximate measure of the risk to particular parts of the body for a given dose equivalent. Some parts are more susceptible to radiation damage (e.g., lungs, bone marrow, or gonads). These have higher risk-weighting factors than other parts of the body. An overall *effective dose equivalent* for the whole body can be calculated from the organ dose equivalents and risk-weighting factors. The annual effective dose equivalent for the average member of the UK population arising from all sources is 2.5mSv, to which exposure of the lungs by radon and its daughters contributes about half. Exposure in the home to a radon gas concentration of 48Bq^m⁻³ would lead, in the course of a year, to an effective dose equivalent of 1mSv (ICRP, 1993). In the United States, the average person is exposed to an effective dose equivalent of approximately 3.6mSv (whole-body exposure) per year from all sources.

Governments set occupational *dose limits* in order to ensure that individuals are not exposed to an unacceptable degree of risk from artificial radiation. Occupational levels are conventionally expressed in working level (WL) units. A WL is any combination of short-lived radon daughters (decay products or progeny ²¹⁸Po, ²¹⁴Pb, ²¹⁴Bi, and ²¹⁴Po) in one liter of air that will result in the emission of 1.3 X 10⁵MeV of potential alpha energy. Exposures are measured in working level months (WLM). A WLM is the cumulative exposure equivalent to 1WL for a working month (170 hours). In SI units, a WLM is defined as 3.54mJhm³ (ICRP, 1993). One WL is approximately equal to a radon exposure of 7500Bq^m⁻³ and 1 WLM to an average radon exposure of about 144Bq^m⁻³ y (on the assumption that people spend most of their time indoors) (NRPB, 2000).

The International Commission for Radiological Protection (ICRP) has recommended that all radiation exposures should be kept as low as reasonably achievable taking into account economic and social factors. In the UK, statutory regulations apply to any work carried out in an atmosphere containing ²²²Rn gas at a concentration in air, averaged over any 24-hour period, exceeding 400Bq^m⁻³ except where the concentration of the short-lived daughters of ²²²Rn in air averaged over any 8-hour working period does not exceed 6.24 X 10⁷Jm⁻³. The limit on effective dose for any employee of 18 years of age or above is 20mSv in any calendar year. This dose limit may be compared with the dose to the average person in the UK of 2.5mSv, the dose of 7.5mSv to the average person living in the high radon area of Cornwall, UK, and 4.5mSv to the average nuclear worker in the UK.

In the United States, exposure limits vary by regulating agency and type of worker. The Miners Safety and Health Act (MSHA) covers underground miners, whereas the Occupational Safety and Health Act (OSHA) regulates exposure to ²²²Rn gas and ²²²Rn progeny for workers other than miners. The MSHA sets limits so that no employee can be exposed to air containing ²²²Rn progeny in excess of 1WL (100pCiL⁻¹) in active work areas. The MSHA also limits annual exposure to ²²²Rn progeny to less than 4 WLM per year. OSHA limits exposure to either 30pCiL⁻¹ or 0.33WL based on continuous workplace exposure for 40 hours per week, 52 weeks per year.

A number of occupations have the potential for high exposure to ²²²Rn progeny: mine workers, including uranium, hard rock, and vanadium; workers remediating radioactive contaminated sites, including uranium mill sites and mill tailings; workers at underground nuclear waste repositories; radon mitigation contractors and testers; employees and recreational visitors of natural caves (Gillmore et al., 2002; Gillmore et al., 2000); phosphate fertilizer plant workers; oil refinery workers; utility tunnel workers; subway tunnel workers; construction excavators; power plant workers, including geothermal power and coal; employees of radon health mines; employees of radon balneotherapy spas (waterborne ²²²Rn source); workers and visitors to hot spring hotels (Song et al., 2011); water plant operators (waterborne ²²²Rn source); fish hatchery attendants (waterborne ²²²Rn source); employees who come in contact with technologically enhanced sources of naturally occurring radioactive materials; and incidental exposure in almost any occupation from local geologic ²²²Rn sources. Recreational and other visitors to abandoned metalliferous mines may also be exposed to high radon concentrations (Gillmore et al., 2001).

Radiation can interact with the electrons in surrounding molecules in the cells and induce changes such as ionization. Ionization of water molecules in organic tissues can alter important molecules in that tissue. Radiation can also ionize and produce chemical changes in DNA molecules, the basic material that controls the structure and function of the cells that make up the human body. This can lead to biological effects, including abnormal cell development, some of which may not be seen for some time after radiation exposure. Alpha particles are considered to be the most dangerous type of radiation. Although they do not penetrate very far, the mass and charge of the particles is so high that it can cause intense ionization. Whereas alpha radiation cannot penetrate the surface layer of the skin (stratum corneum), the interior of the lungs lacks a protective epidermis so that alpha decay particles emitted by radon progeny can damage important molecules in the cells. Gamma rays are very penetrative and can cause ionization and tissue damage comparable in effect to x-radiation, but are usually much more energetic. Provided the radioactive sources remain outside the body, gamma radiation is the greatest problem because it is so penetrating, whereas alpha particles are stopped by clothing and the outer layers of the skin. Beta particles are intermediate in penetrating power.

When radioactive sources are taken into the body however, the situation changes markedly. The major pathways by which alpha activity enters the human body are the ingestion of radioactive elements and inhalation of radon, and more importantly its daughter products, some of which are alpha particle emitters (^{218}Po and ^{214}Po). Alpha particles give up their energy to a very small volume of tissue and can thus cause intensive damage, which has been shown to result in cancers. Much of the inhaled radon is exhaled and relatively few alpha particles are emitted by it within the body. However, the four immediate decay products of ^{222}Rn have short half-lives and are all radioactive isotopes of solid elements (Table IV). The decay products, which remain in suspension attached to the surface of aerosols, dust, smoke, or moisture particles, or are unattached, may remain in the respiratory system where they may become trapped in the lungs and irradiate the cells of mucous membranes, bronchi, and other pulmonary tissues. Overall doses are due largely to irradiation of the bronchial epithelium and secretory cells by alpha particles from the short-lived decay products of ^{222}Rn . It is believed that the ionizing radiation energy affecting the bronchial epithelial cells initiates carcinogenesis. As a consequence the main danger is an increased risk of developing cancers of the respiratory tract, especially the lungs. Whereas radon-related lung cancers occur primarily in the upper airways, radon increases the incidence of all histological types of lung cancer, including small cell carcinoma, adenocarcinoma, and squamous cell carcinoma. The contribution to both lung dose equivalent and effective dose equivalent by the beta and gamma radiations may be ignored, as they are small compared to those from alpha radiation.

It is interesting to note that radon remained a chemical curiosity for decades, even promoted at times as a "health-giving" gas at various spas. Initially radon was regarded as a fairly innocuous or even benign component of geological gases, and its importance as the major contributor to the radioactive dose received by the general population has been recognized relatively recently. In contrast to the early dramatic effects of high radiation doses on humans, which can cause death in a few days or weeks, or obvious skin damage when a limited area of the body is exposed to a high radiation dose, the effects of the relatively low doses of natural radiation (e.g., cancer) usually occur a long time after exposure.

The overall hazard to human health from gamma radiation, either indoors or outdoors, is negligible compared with the hazard associated with radon.

IV. Radon Epidemiology

Evidence linking the exposure to high levels of radon and an increase in the risk of lung cancer is becoming overwhelming. Indeed more is known about the health risks of radon exposure than about most other human carcinogens. A large body of epidemiological data has accumulated over several decades relating to studies of the incidence of lung cancer in miners and risk estimates have been derived from these data (NAS, 1998; NRPB, 2000). Supporting evidence comes from experimental studies of animals and from radiobiology.

A. Cohort Studies of Miners

High death rates from lung cancer recorded in the Middle Ages among miners in Germany and the Czech Republic are now recognized as radon-induced. Studies of thousands of miners, some with follow-up periods of more than thirty years, have been conducted in uranium, iron, tin, and fluor spar mines in Australia, Canada, China, Europe, and the United States. These studies consistently demonstrated an increase in lung cancer incidence with exposure to radon decay products, despite several differences in study populations and methodologies. The miner studies demonstrated that (1) at equal cumulative exposures, low exposures in the range of the U. S. Environmental Protection Agency's (EPA) 4pCiL^{-1} ($148\text{Bq}\text{m}^{-3}$) action level over longer periods produced greater lung cancer risk than high exposures over short periods; (2) increased lung cancer risk with radon exposure was observed even after controlling for, or in the absence of, other potentially confounding mine exposures such as asbestos, silica, diesel fumes, arsenic, chromium, nickel, and ore dust; (3) increased lung cancer risk has been observed in miners at relatively low cumulative exposures in the range of the U. S. EPA's 4pCiL^{-1} ($148\text{Bq}\text{m}^{-3}$) action level; and (4) nonsmoking miners exposed to radon have been observed to have an increased risk of lung cancer.

A major reassessment of health information mainly on uranium miners from Czechoslovakia, Colorado, Sweden, and Canada by the IRCP (1993) demonstrated a pronounced excess of lung cancers. More recently the Sixth Committee on the Biological Effects of Ionizing Radiation (BEIR VI) of the American National Academy of Sciences (NAS, 1998) re-analyzed the data for miner cohorts and used the most recent information available to estimate the risks posed by exposure to radon in homes.

Results from twelve major epidemiological studies involving a total of more than 60,000 miners clearly indicate a correlation between excess mortality from lung cancer and radon exposure (Figure 1). A combined analysis of studies on underground miners revealed an increase in relative risk from about 2% at a mean exposure of 250 WLM to 10% at 2500 WLM (Lubin et al., 1994).

Differences between mine and home environments could limit the validity of extrapolating risk estimates based on miner data to the home environment. These include generally higher ^{222}Rn gas concentrations, more airborne dust, and larger dust particle diameter in mines compared with homes as well as different activity size distributions of radon progeny and rates of attachment. Other toxic pollutants present in mine air may act as confounders. In addition, there are age and sex differences between miners and the general population; higher levels of physical activity among miners, which affects respiration rates; and more oral (as opposed to nasal) breathing in the miners, which leads to increased deposition of larger particles into the lung. Miners have shorter term high exposure compared with the lifelong lower concentration exposure for the general population. Finally, most miners were smokers compared with a minority of the general population. Even allowing for these factors, the evidence for a causal association between lung cancer and occupational radon exposure in underground miners is overwhelming (see also Chapter 12, this volume).

B. Case Control Studies of the Effects of Domestic Exposure

Comparison of radon exposures among people who have lung cancer with exposures among people who have not developed lung cancer is the most direct way to assess the risks posed by radon in homes. Many factors must be considered when designing a domestic case control radon epidemiology study. These factors include

1. Mobility: People move a lot over their lifetime and it is virtually impossible to test every home where an individual has lived; estimates of radon exposure have to be used to fill in gaps in exposure history.
2. Housing stock changes: Over time, older homes are often destroyed or reconstructed so that radon measurements will be not available or vary dramatically from the time of occupancy by the case; a home's radon level may increase or decrease over time if new ventilation systems are installed, the occupancy patterns may change substantially, or the home's foundation may shift and cracks appear.

3. Inaccurate histories: Often a majority of the lung cancer cases (individuals) studied are deceased or too sick to be interviewed by researchers. This requires reliance on second-hand information, which may not be accurate.

These inaccuracies primarily affect

1. Residence history: A child or other relative may not be aware of all residences occupied by the patient, particularly if the occupancy is distant in time or of relatively short duration. Even if the surrogate respondent is aware of a residence he or she may not have enough additional information to allow researchers to locate the home.
2. Smoking history: Smoking history historically has reliability problems. Individuals may underestimate the amount they smoke. Conversely, relatives or friends may overestimate smoking history.
3. Other factors: Complicating factors other than variations in smoking habits include an individual's genetics, lifestyle, exposure to other carcinogens, and home heating, venting, and air conditioning preferences.

Several such case control (or cohort) studies have been completed but they have not produced a definitive answer, principally because the risk is small at the low exposure of most domestic environments. In addition, many people involved in the studies moved a number of times so it was difficult to estimate the radon exposures that people had received over their lifetimes. The greatest problem, however, was caused by the fact that far more lung cancers are caused by smoking than are caused by radon (NAS, 1998).

Residential epidemiological case-control studies examining the relationship between contemporary ^{222}Rn gas concentrations in homes and lung cancer have been performed in Canada, China, Finland, Germany, Sweden, the UK, and the United States. These studies indicate that higher lung cancer rates occur in those exposed to higher levels of radon, although in most cases this did not reach a statistical level of significance (Lubin & Boice, 1997; Darby et al, 1998).

Meta-analysis is a statistical attempt to analyze the results of several different studies to assess the presence or absence of a trend or to summarize results. Meta-analysis of the largest case control studies produced a positive risk estimate that was statistically significant and close to that derived from the miner data (Lubin & Boice, 1997). The meta-analysis of eight studies using weighted linear regression found a summary excessive risk of 14% at an average indoor ^{222}Rn gas concentration of 4pCiL^{-1} (148Bqm^{-3}). The excess risk at 4pCiL^{-1} (148Bqm^{-3}) in recent studies in Germany and the UK was in close agreement with risk estimates obtained from the meta-analysis. Lubin and Boice (1997) concluded that the results of their meta-analysis are consistent with the current miner-based estimates of lung cancer risk from radon, which place the number of radon-related deaths at approximately 15,000 per year in the United States. Because meta-analysis has several inherent limitations (such as the inability to adequately explore the consistency of results within and between studies and to control for confounding factors), meta-analysis is not able to prove that residential radon causes lung cancer. But it does provide additional good suggestive evidence. It is one more link in the "chain of evidence" connecting residential radon exposure to increased lung cancer risk. Because the investigators performing a meta-analysis do not have access to the raw data on the individual study subjects, the analysis is based on the published relative risks and confidence intervals of the individual studies. Frequently, the impact of each study is weighted based on some factor considered relevant to the reliability of each study's data. In the Lubin and Boice (1997) meta-analysis, the results of each individual study were weighted so that each study contributed in relation to the precision (i.e., relative lack of random or sampling errors) of its estimate.

An exposure-rate effect is the alteration of an effect by intensity of an exposure. An inverse exposure-rate effect would be the enhancement of an effect as the intensity of the exposure decreases (i.e., low-level chronic exposures would be riskier than high-level more acute exposures). An "inverse exposure-rate effect" was observed in the miner data. This means that for miners who received the same exposure, those that received it over a longer period of time had a greater risk of lung cancer. The inverse exposure-rate effect diminished in miners exposed below 50–100 WLM. The finding that the inverse exposure-rate effect does not seem to apply in residential situations will not change the EPA's risk assessment since the EPA had not included the inverse exposure-rate effect in their latest, 1992, risk estimate.

Collaborative analysis of individual data from 13 case-control studies of residential radon and lung cancer in 9 European countries showed that the risk of lung cancer increased by 16% (95% confidence interval = 5% to

31%) per 100 Bq m⁻³ increase in radon after correction for random uncertainties in measuring radon concentrations (Darby et al., 2005). The dose-response relation seemed to be linear with no threshold and remained significant (P=0.04) in analyses limited to individuals from homes with measured radon < 200 Bq m⁻³. The results suggest that radon is responsible for about 2% of all deaths from cancer in Europe. Systematic analysis of pooled data from 7 North American residential case-control radon studies showed that the odds ratios (ORs) for lung cancer increased with residential radon concentration. The estimated OR after exposure to radon at a concentration of 100 Bq m⁻³ over a 25 yr period was 1.11 (1.00-1.28), which is compatible with the estimate of 1.12 (1.02-1.25) predicted by downward extrapolation of the results from occupational studies of radon-exposed underground miners (Krewski et al., 2005; Krewski et al., 2006).

C. Ecological (Geographical) Epidemiological Studies

Ecological epidemiological studies of the associations between average lung cancer rates and radon concentrations in geographical areas are considered to be much less reliable than case-control studies that consider individual radon exposure and smoking histories. A negative correlation between mean radon and lung cancer rates in counties in the United States (Cohen, 1997) is not well understood and the study methodology has been criticized by epidemiologists (e.g., Lubin, 1998; Smith et al., 1998). Ecological studies of the association between childhood cancer and indoor radon exposure are reviewed in (HPA, 2009) (see also Chapter 21, this volume).

D. Extrapolation from Mines to Homes

Whereas it has been suggested that the dose response relationship seen in miners may not extend to the much lower levels present in most homes, there appears to be sufficient evidence to suggest that a dose threshold for radiation carcinogenesis does not apply to lung cancer. Ionizing radiation is thought to induce specific gene mutations in DNA in single target cells in tissue and, as such, act principally at the initial stage of cancer. The number of cells hit by alpha particles will be broadly proportional to the dose (i.e., the radon concentration in a dwelling). The general consensus is that low dose (i.e., domestic radiation) cancer risk rises in proportion to the dose and there is not a threshold below which risk may be discounted (NRPB, 2000).

More information about residential exposure to radon is needed to answer important questions about radon's effect on women and children—groups not included in the occupational studies of miners. Although children have been reported to be at greater risk than adults for developing certain types of cancer from radiation, currently there is no conclusive evidence that radon exposure places children at any greater risk. Some miner studies and animal studies indicate that for the same total exposure, a lower exposure over a longer time is more hazardous than short, high exposures. These findings increase concerns about residential radon exposures.

E. Experimental Studies with Animals

Results from animal experiments conducted in the United States and France are generally consistent with the human epidemiological data. Health effects observed in animals exposed to radon and radon decay products include lung carcinomas, pulmonary fibrosis, emphysema, and a shortening of life span (USDOE, 1988). The incidence of respiratory tract tumors increased with an increase in cumulative exposure and with a decrease in rate of exposure (NAS, 1998). Increased incidence of respiratory tract tumors was observed in rats at cumulative exposures as low as 20WLM (NAS, 1998). Exposure to ore dust or diesel fumes simultaneously with radon did not increase the incidence of lung tumors above that produced by radon progeny exposures alone (USDOE, 1988). Lifetime lung tumor risk coefficients observed in animals are similar to the lifetime lung cancer risk coefficients observed in human studies (USDOE, 1988). In a study of rats simultaneously exposed to radon progeny and uranium ore dust, it was observed that the risk of lung cancer was elevated at exposure levels similar to those

found in homes. The risk decreased in proportion to the decrease in radon-progeny exposure (Cross, 1992). Confounding factors such as smoking are more readily controlled in animal experiments and qualitatively confirmed that radon can indeed induce lung cancer in the absence of smoking. Animal evidence on radon and lung cancer is reviewed in HPA (2009)(HPA, 2009).

F. Other Cancers and Radon Exposure

No consistent association has been observed between radon exposure and other types of cancer. A combined analysis of the data from 11 miner cohort studies involving more than 60,000 miners did not find convincing links (Darby et al., 1995). No clear association between radon and childhood cancer (especially leukemia) emerged from a number of ecological studies, and a review of ecological, miner, and cohort studies did not find an association between radon and leukemia (Laurier et al., 2001).

Radon ingested in drinking water may lead in some circumstances to organs of the gastrointestinal tract receiving the largest dose. Ingested radon is absorbed by the blood. Most of the radon is lost quickly from the bloodstream through the lungs but some will deliver a dose to other body organs, especially those with a high fat content due to the higher solubility of radon in fat compared with water. Other body organs may be irradiated to some extent although the doses involved will be much smaller. Comparative estimated doses to various organs from exposure to radon are indicated in **Table V**. Apart from lung cancer, there is no epidemiological proof of radon causing any other type of cancer. If radon and its decay products have an effect on organs other than the lungs, this is too weak to be readily detectable (HPA, 2009)(HPA, 2009).

V. Radon Health Risks

Because a valid risk estimate could not be derived only from the results of studies in homes, the BEIR VI committee chose to use data from studies of miners to estimate the risks posed by radon exposures in homes (NAS, 1998). The committee statistically analyzed the data from 11 major studies of underground miners, which together involved about 68,000 men, of whom 2700 have died from lung cancer. A range of models was used to try to explain the relationship between radon and smoking. In the multiplicative model it is assumed that a specific radon exposure will multiply the base risk rate for smokers and non-smokers by the same factor. BEIR VI models take into account total exposure, age and duration of exposure or total exposure, and age and average radon exposure with predicted risks at about 50% higher under the first of these two models. In general, the risk of developing lung cancer increases linearly as the exposure increases.

Epidemiological studies confirm that exposure to radon in homes increases the risk of lung cancer in people. It is estimated that between 3% and 14% of all lung cancers are linked to radon. Radon is the second most important cause of lung cancer after smoking and is much more likely to cause lung cancer in people who smoke, or who have smoked, than in lifelong non-smokers for whom radon is the primary cause of lung cancer (WHO, 2009).

A study of lung cancer deaths from indoor radon in the UK (HPA 2009) estimated that: (i) 1100 lung cancer deaths were caused by radon, representing 3.3% of total lung cancer deaths (34 000); (ii) the dose-response relationship appears linear, in that the greater the concentration of indoor radon, the greater the risk of developing lung cancer; (iii) there is no evidence of any threshold below which there is no risk; (iv) of the 3.3% of lung cancer deaths attributable to radon exposure, only 0.5% were due to radon acting alone: the remaining 2.8% were caused by a combination of radon and smoking, with nearly half the deaths likely to occur in people who had already given up smoking. The HPA research also highlighted that the vast majority of radon-induced lung cancer deaths occur in areas that are not currently designated as 'Radon Affected Areas' (areas in which over 1% of homes are estimated to exceed 200 Bq m⁻³, the UK radon Action Level). The overwhelming majority of the popu-

lation live outside such areas, and around 95% of radon-attributable deaths are estimated to occur with residential concentrations below 200 Bq m^{-3} , and 70% at concentrations of less than 50 Bq m^{-3} . This has major implications for the cost-effectiveness of government intervention strategies designed to manage exposure to radon in the domestic environment (HPA 2009; WHO 2009).

The number of lung cancer cases due to residential radon exposure in the United States was estimated to be 15,400 (exposure-age-duration model) or 21,800 (exposure-age-concentration model), which is 10–15% of lung cancer deaths. Radon causes 11% of lung cancer deaths among smokers (most of whom die of smoking) but 23% of never-smokers. The BEIR VI committee's uncertainty analyses using the constant relative risk model suggested that the number of lung cancer cases could range from about 3000 to 32,000. The 95% upper confidence limit for the exposure-age-concentration model was approximately 38,000, but it was considered that such a high upper limit was highly unlikely given the uncertainty distributions. The major shortcomings in the existing data relate to estimating lung cancer risks near 148 Bq m^{-3} (4 pCi L^{-1}) and down to the average U. S. indoor level of 46 Bq m^{-3} (1.24 pCi L^{-1}), especially the risks to never-smokers.

Most of the radon-related deaths among smokers in the United States would not have occurred if the victims had not smoked. Whereas there is evidence for a synergistic interaction between smoking and radon, the number of cancers induced in ever-smokers by radon is greater than one would expect from the additive effects of smoking alone and radon alone. The estimated number of deaths attributable to radon in combination with cigarette smoking and radon alone in never-smokers constitutes a significant public-health problem and makes indoor radon the second leading cause of lung cancer after cigarette smoking.

If the radon risk in China and India is approximately the same as in the UK, then, based on similar average indoor radon concentrations (Cheng et al., 2002; Ramachandan and Sathish, 2011), radon may cause over 20,000 deaths per year in each of these two countries. The percentage of lung cancer deaths attributed to radon is 7.8% in Canada, 5% in Germany, 8.3% in Switzerland, and between 5 and 12% in France (WHO, 2009).

The BEIR VI committee suggested that about one-third of the radon-attributed cases (about 4% of the total lung cancer deaths) would be avoided if all homes had concentrations below the EPA's action guideline of 148 Bq m^{-3} (4 pCi L^{-1}). Of these deaths, about 87% would be in ever-smokers. Deaths from radon-attributable lung cancer in smokers could be reduced most effectively through reduction in smoking, because most of the radon-related deaths among smokers would not have occurred if the victims had not smoked. Whereas the relative risks for smokers and nonsmokers is still disputed, evidence from miners who never smoked demonstrates a clear relationship between cumulative exposure and relative risk. Existing biologic evidence indicates that even very low exposure to radon might pose some risk but that a threshold level of exposure, below which there is no effect of radon, cannot be excluded.

BEIR VI risk models have been used to estimate fatal lifetime lung cancer risk for lifetime exposure at 200 Bq m^{-3} (Table VI). More recently, the UK HPA used risk estimates from the European pooling study ((Darby et al., 2005) to calculate the cumulative absolute risk of death from lung cancer to age 75 years in the UK by long-term average residential radon concentrations and smoking history (Table VII). Combining the results of the European, North American and Chinese pooling studies gives an estimated increased risk of lung cancer of 10% per 100 Bq m^{-3} increase in measured radon concentration (WHO, 2009).

At the mean domestic radon concentration of 21 Bq m^{-3} , the risk for a continuing cigarette smoker is more than 30 times greater than for a lifelong non-smoker. The risk is about 30% higher in houses with radon concentrations at the current UK action level of 200 Bq m^{-3} . To put this in perspective, risk of death from accidents in the home is 0.7%; risk of premature death from accidents on the road is 2.5%, while there is a 5–7.5% overall risk of lung cancer in the UK.

Duport (2002) questioned whether radon risk is overestimated because only the exposure to inhaled radon decay products is generally taken into account in the determination of risk of radiogenic lung cancer in uranium miners, whereas the risk actually reflects the total dose of radiation received by the lung. Radiation dose from sources other than ^{222}Rn decay products may account for 25–75% of the total effective dose, absorbed dose, or equivalent lung dose and this varies between mines. Neglecting these doses would lead to overestimation of risk both through dose underestimation and misclassification. Correction for neglected doses and dose misclassification would reduce the risk per unit of radon exposure by a factor of at least two or three and bring the overall

dose-effect relationship toward the no-effect null hypothesis. This would increase the likelihood of a radon exposure threshold for lung cancer risk at current indoor exposure levels (Duport, 2002).

The U. S. EPA estimates that radon in drinking water causes about 168 cancer deaths per year, 89% from lung cancer caused by breathing radon released from water, and 11% from stomach cancer caused by drinking radon-containing water. In general, radon released from tap water and inhaled will present a greater risk than radon ingested through drinking water (NRPB, 2000)(HPA, 2009).

Estimation of the economic cost of radon-induced lung cancers is difficult. Using the Quality Adjusted Life Years approach,(Gray et al., 2009) the total estimated cost of radon-induced lung cancers is £347 million per year in the UK.

VI. Sources of Natural Radiation

A. Introduction

The average person in the UK receives an annual effective radiation dose of 2.8mSv, of which about 85% is from natural sources: cosmic rays, terrestrial gamma rays, the decay products of ^{220}Rn and ^{222}Rn , and the natural radionuclides in the body ingested through food and drink. Of these the major proportion is from geological sources. About 60% of the total natural radiation dose is from radon isotopes (mostly due to alpha particle activity) while about 15% is thought to be due to gamma radiation from the U, Th, and K in rocks and soils and from building products produced from geological raw materials (Akerblom et al., 2005). X-rays and radioactive materials used to diagnose disease are the largest source of artificial exposure to people. The average dose due to anthropogenic isotopes (radioactive fallout, fuel cycle, etc.) is less than 1% of the total annual dose (Figure 2). Similar average annual effective doses apply worldwide (UNSCEAR, 2009). In European countries, the average annual dose from natural sources is 2mSv in Denmark and the UK rising to 3mSv in Finland and Sweden where indoor radon concentrations and gamma radiation are much higher (NRPA, 2000).

On an individual basis, the dose would be dependent upon where one lived, one's lifestyle, and the nature and extent of any medical treatment. Most of the exposures to terrestrial gamma rays and to ^{220}Rn and ^{222}Rn decay products result from living indoors. Building materials are the main source of thoron (^{220}Rn) in room air although a minor contribution comes from soil gas. Radon contributes by far the largest variation in the average dose from natural radiation sources.

B. Radon

The average annual dose to the UK population from radon is 1.2mSv with a range of 0.3 to more than 100mSv. In the most radon-prone area in Great Britain, the average person receives a total annual radiation dose of 7.8mSv of which 81% is from radon. The production of radon by the radioactive decay of uranium in rock, overburden, and soil is controlled primarily by the amount of uranium within the rock-forming minerals and their weathering products. The ^{238}U decay chain may be divided into two sections separated by ^{226}Ra (radium), which has a half-life of 1622 years (Table I). Earlier isotopes mostly have long half-lives, while the later isotopes, including radon (^{222}Rn), have relatively short half-lives. Outdoors, radon normally disperses in the air whereas in confined spaces such as buildings, mines, and caves (Gillmore et al., 2001; Gillmore et al., 2002; Gillmore et al., 2000) it may accumulate. Higher radon (average 480 Bq m^{-3}) was found in houses located $\pm 150\text{m}$ from the surface projection of a closed uranium mine tunnel in Hungary compared with houses located further away (average 291 Bq m^{-3}) (Somlai et al., 2006). Radon is thought to migrate through fissures that intersect the mine tunnel and run up to the surface. Radon in indoor air comes principally from soil gas derived from soils and rocks beneath a building with smaller amounts from the degassing of domestic water into the indoor air and from building materi-

als. Soil gas represents the predominant source of indoor ^{222}Rn gas. Outdoor radon concentrations may occasionally reach potentially hazardous levels. For example, air escaping from an open uranium mine gallery in the town of Schneeberg, Germany, contained up to $10,000\text{Bq m}^{-3}$ radon and ventilation facilities had to be installed to prevent ingress of this air into an adjacent factory. Radon is potentially emitted from some anthropogenic sources, such as near-surface radioactive waste disposal sites (Appleton et al., 2011a). Radon exhalation and releases from uranium mining and milling has the potential to increase long-term radon releases into the adjacent environment relative to pre-mining baseline concentrations (Mudd, 2008). Precipitation and soil moisture can influence radon flux densities from uranium mining waste rock dumps, ore stockpiles and where effluent enriched in ^{226}Ra has been spray irrigated over land in wet-dry tropical regions (Lawrence et al., 2009).

Water in rivers and reservoirs usually contains very little radon so homes that use surface water do not have a radon problem from their water. Water processing in large municipal systems aerates the water, which allows radon to escape and also delays the use of water until most of the remaining radon has decayed.

Radon dissolved in groundwater can migrate over long distances along fractures and caverns depending on the velocity of fluid flow. Mains water supplies pass through treatments which tend to remove radon gas but small public water works and private domestic wells often have closed systems and short transit times that do not remove radon from the water or permit it to decay. In such situations, radon from the domestic water released during showering and other household activities could add radon to the indoor air. Areas most likely to have problems with radon from domestic water supplies include those with high levels of uranium in the underlying rocks. This association has been observed in the USA, the UK and Sweden (Akerbom et al, 2005; Skepperstrom and Olofsson, 2006).

Radon from degassing of domestic water accounts for about 5% of the total indoor radon for homes that use groundwater sources in the United States. In some cases, radon from this source may account for a higher proportion of indoor radon. In Maine (U. S.), radon concentrations in domestic water wells sometimes exceed $37,000\text{Bq L}^{-1}$ and more than 10% of private well water supplies exceed the action level (740Bq L^{-1}). Radon from the domestic water supply is inhaled when it is released from the water during showering (Song et al., 2011), washing clothes, and washing dishes. It is estimated that 370Bq L^{-1} of radon in the domestic water supply contributes about 37Bq m^3 to the indoor air of a home. Areas most likely to have problems with radon from domestic water supplies include those with high levels of uranium in the underlying rocks, such as uraniumiferous granites. This association has been observed both in the United States and the UK. In Maine, for example, the average for well water in granite areas is over 500Bq L^{-1} . Approximately 10% of the drilled wells in population-based random study of 472 private wells in Finland exceeded a radon concentration of 1000Bq L^{-1} (Vesterbacka et al., 2005) whilst in Sweden, 10,000 private wells are estimated to have concentrations greater than 100Bq L^{-1} , which is the recommended action levels for private households.

In a study of private water supplies in southwestern England, a high proportion derived from granite areas exceeds the draft European Union action level of 1000Bq L^{-1} . It was also found that radon concentrations varied significantly over the course of a week and between samples taken several months apart. For water from groundwater sources, mean values (by source type) at the tap were generally lower than those at the source. This is consistent with loss of radon due to degassing as a result of water turbulence within the supply system and natural radioactive decay while the water is resident in the household supply system. All the water sources sampled showed large variability in radon concentration over the summer sampling period, whereas less pronounced variability was observed during the winter sampling. Maximum values were observed during the summer.

In SW Poland there is a correlation between radon potential and radon in groundwater where high concentrations are associated with enhanced levels of ^{226}Ra , high emanation coefficients, intense rock fracturing and intensive weathering of bedrocks (Przylibski et al., 2004). In the Extremadura region of Spain, the highest radon in groundwater ($>1000\text{Bq L}^{-1}$) occurs in areas of granitic and Precambrian igneous bedrock, where there is a positive correlation between ^{222}Rn in groundwater and natural gamma radiation (Galán López and Martín Sánchez, 2008).

Building materials generally contribute only a small percentage of the indoor air ^{222}Rn concentrations, although this may be 20-50% of the radon in an average UK dwelling (Gunby et al., 1993), Groves-Kirby et al. (2008) (Groves-Kirby et al., 2008) review this topic and noted that in areas of generally low indoor radon concentrations, indoor radon may be mainly derived by emanation from building materials, with an estimated 20, 10-70, and 20, Bq m^{-3} of indoor radon being derived from this source in Belgium, Germany and the Netherlands respectively. However, in some areas, concrete, blocks, or wallboard made using radioactive shale or waste products from uranium mining will make a larger contribution to the indoor radon. High radium content and radon exhalation rates in concrete and bricks used in some Hong Kong buildings with high indoor radon concentrations are probably caused by the granitic composition of aggregates (Man & Yeung, 1998). In Sweden, 300,000 houses constructed with radioactive alum shale form the world's largest stock of buildings that have used building materials with enhanced radiation (Akerblom et al., 2005). The houses have radon concentrations of 100–400 Bq m^{-3} and gamma radiation levels of 0.3–1.2 $\mu\text{Sv h}^{-1}$ (NRPA, 2000). High effective dose rates (7.1–16.7 mSv y^{-1}) for ^{222}Rn and ^{220}Rn and their progenies have been estimated for cave dwellings excavated in loess in the Yan'an area of China, reflecting both exhalation from the loess and relatively poor ventilation (Weigand et al., 2000). More recent radon and thoron measurements in loess cave dwellings in Gansu province, China revealed arithmetic mean radon and thoron concentrations of 91 and 351 Bq m^{-3} , respectively producing an effective dose of 3.08 mSv/yr , significantly lower than earlier estimates (Yamada et al., 2006). Geology, coating level and material, decorating material and ventilation were concluded to be the main factors affecting radon concentration in underground buildings in China (Li et al., 2006). Tunnels built in granite, tuff, quartz sandstone and limestone generally have higher radon concentrations in summer compared with winter, although the reverse was observed in andesitic porphyry tunnels. However, the difference was statistically significant only in limestone tunnels (Li et al., 2010). Granite derived soil used for building materials in Guangdong Province, China, has high ^{238}U (140 Bq kg^{-1}) and ^{226}Ra (134 Bq kg^{-1}) producing radium equivalent activities much higher than the national limit value for building material (Song et al., 2012).

C. Gamma Rays From the Ground and Buildings (Terrestrial Gamma Rays)

Everyone is irradiated by gamma rays emitted by the radioactive materials in the Earth. Building materials extracted from the Earth may also emit gamma radiation, so people can be irradiated indoors as well as outdoors. Terrestrial gamma rays originate principally from the radioactive decay of the natural potassium, uranium, and thorium. These elements are widely distributed in terrestrial materials which include rocks and soils. The average annual gamma radiation dose from all these sources to the population in Great Britain is about 350 μSv with a range of 120–1200 μSv .

In general, the gamma radiation dose is proportional to the amount of U, Th, and K in the ground and in building materials. Within a masonry building, most of the gamma radiation is received from the building materials, whereas in wooden buildings a larger part of the dose is contributed from gamma radiation from the ground. The average person in the UK spends only 8% of his or her time outdoors so the contribution to total radiation dose from the ground is relatively small. The bulk of the radiation above the ground surface is derived from only the top 30cm or so of soil or rock. Soils developed upon radioactive rocks generally have a much lower gamma radioactivity than the rock substrate. Whereas one can predict or identify areas of high geological gamma radioactivity, the resultant dose to the population depends on additional factors such as soil type, house construction, and life-style.

A single K isotope, ^{40}K , comprising only 0.0119% of the total K, is radioactive. ^{40}K undergoes branched decay producing ^{40}Ca and ^{40}Ar , the latter reaction producing high-energy gamma rays. Uranium consists of two main isotopes (^{238}U and ^{235}U). Because ^{235}U comprises only 0.72% of the total U it may, for practical purposes, be ignored. Isotopes in the later section of the ^{238}U decay series (Table I) include the bismuth isotope (^{214}Bi), which contributes most of the gamma activity of the decay series. As ^{226}Ra is chemically very different from U, it is possible in natural processes for the two to become separated so that the ^{226}Ra , and its daughter products are unsupported by the parent U. In addition, radioactive elements in the rock fragments and derived

minerals in the weathered overburden are diluted with organic matter and water. Thus there may not be a simple relationship between the measured gamma ray flux and the U content.

Many of the daughter isotopes of ^{238}U and ^{232}Th are gamma active. ^{208}Tl is the main gamma active daughter product derived from ^{232}Th , and this takes 70 years to reach secular equilibrium in the ^{232}Th decay series. Potassium gives rise to a prompt gamma ray in which the intensity is directly related to the potassium concentration. On average potassium is much more abundant than thorium which, in turn, is more abundant than uranium. However, the specific gamma activities are such that, on average, approximately equivalent gamma emissions are observed from potassium and the decay series of uranium and thorium.

Areas of high natural radiation include areas of monazite sands in Brazil, China, Egypt, and India; volcanic rocks in Brazil and Italy; uranium mineralization in France, the UK, and the United States; and radium-enriched karst soils developed over limestones in Switzerland, the UK, and the United States (UNSCEAR, 2009).

D. Food and Drink

Radioactive materials even occur in food. Potassium-40 in particular is a major source of internal irradiation. Natural radioactivity in the human diet gives an average annual dose of $300\mu\text{Sv}$ each year of which $180\mu\text{Sv}$ is from ^{40}K . The range for all internal radiation sources in Great Britain is $100\text{--}1000\mu\text{Sv}$ per annum. Shellfish concentrate radioactive materials so that, even when there is no man-made radioactivity, people who consume large quantities of mussels, cockles, or winkles can receive a dose from natural radioactivity in food that is about 50% higher than average. Apart from restricting intake of shellfish, there is very little possibility of reducing the small exposure to natural radioactivity from food.

E. Cosmic Rays

Approximately 10% of the average annual radiation dose is from cosmic rays (Table VIII), although this increases with latitude and altitude. The average dose from cosmic radiation received each hour rises from $0.03\mu\text{Sv}$ at sea level, to $0.1\mu\text{Sv}$ in Mexico City (altitude of 2250m), and $5\mu\text{Sv}$ at the cruising altitude for commercial jet aircraft (10,000m). Polar and mountain dwellers, aircrews, and frequent air travelers therefore receive higher doses of cosmic radiation. Little can be done about cosmic radiation because it readily penetrates ordinary buildings and aircraft. The average annual dose from cosmic rays in Great Britain is $250\mu\text{Sv}$, with a range of $200\text{--}300\mu\text{Sv}$.

VII. Measurement of Radon

A. Radon Testing in the Home

Although radon cannot be seen or smelled, it can be measured relatively easily with the proper equipment. The most common procedures for measuring radon make use of the fact that it is the only natural gas that emits alpha particles, so if a gas is separated from associated solid and liquid phases any measurements of its radioactive properties relate to radon or its daughter products. In the United States, radon in homes is usually measured using inexpensive do-it-yourself radon test kits, which are available by mail order and in many retail outlets or by hiring a U. S. EPA qualified or state-certified radon tester. The EPA recommends that all homes be tested for radon below the third floor.

Common short-term test devices are charcoal canisters, alpha track detectors, liquid scintillation detectors, electret ion chambers, and continuous monitors. A short-term testing device remains in the home for 2–90 days, depending on the type of device. Because radon levels tend to vary from day-to-day and season-to-season, a long-term test is more likely than a short-term test to measure the home's year-round average radon level. If results are needed quickly, however, a short-term test followed by a second short-term test may be used to deter-

mine the severity of the radon problem. Long-term test devices, comparable in cost to devices for short-term testing, remain in the home for more than three months. A long-term test is more likely to indicate the home's year-round average radon level than a short-term test. Alpha track detectors and electret ion detectors are the most common long-term test devices.

Charcoal canister and liquid scintillation detectors contain small quantities of activated charcoal. Radon and its decay products are adsorbed onto the charcoal and are measured by counting with a sodium iodide detector or a liquid scintillation counter. Radon adsorbed at the beginning of the exposure decays away after a few days so the duration of the measurement is restricted and the device does not measure the true average exposure. Charcoal detectors are suitable only for short-term tests when results are required urgently and a less accurate measurement is acceptable. The result should be well below the action level before it can be concluded that the annual average concentration will also be below the action level. Ambiguous short-term measurements should be followed up by a long-term measurement (NRPB, 2000).

Alpha (etched) track detectors contain a small sheet of plastic that is exposed for a period of one to three months. Alpha particles etch the plastic as they strike it. These marks are then chemically treated and are usually counted automatically under a microscope to determine the radon concentration. Etched track detectors are relatively cheap and suitable for long-term measurement and are usually deployed for a period of three months.

Electret ion detectors contain an electrostatically charged Teflon disk. Ions are generated by the decay of radon strike and reduce the surface voltage of the disk. By measuring the voltage reduction, the radon concentration can be calculated. Allowance must be made for ionization caused by natural background radiation. Different types of electret are available for measurements over periods of a few days to a few months. The detectors must be handled carefully for accurate results.

Continuous monitors are active devices that need power to function. They require operation by trained testers and work by continuously measuring and recording the amount of radon in the home. These devices sample the air continuously and measure either radon or its decay products (NRPB, 2000).

A rigorous procedure must be followed for short-term tests if relatively reliable results are to be obtained. For example, doors and windows must be closed 12 hours prior to testing and throughout the testing period. The test should not be conducted during unusually severe storms or periods of unusually high winds. The test kit is normally placed in the lowest lived-in level of the home, at least 50cm above the floor, in a room that is used regularly, but not in the kitchen or bathroom where high humidity or the operation of an exhaust fan could affect the validity of the test. At the end of the test period, the kit is mailed to a laboratory for analysis; results are mailed back in a few weeks. If the result of the short-term test exceeds 100Bqm^3 then a long-term test is normally recommended.

Remediation of the home is recommended if the radon concentration exceeds certain levels (150Bqm^{-3} in Luxembourg and the United States; 200Bqm^3 in Australia, Israel, Syria, and the UK; 400Bqm^{-3} in Austria, Belgium, Denmark, Finland, Greece, and Sweden).

Radon levels are generally highest in winter (e.g. (Bossey and Lettner, 2007) so seasonal corrections are usually applied to estimate the average annual radon level. In workplaces, consideration needs to be taken of work practices and the building design and use. For small premises at least one measurement should be made in the two most frequently occupied ground floor rooms. In larger buildings at least one measurement is required for every 100m^2 floor area. Regional variation of seasonal correction factors have been observed in Ireland and some regions in the UK have seasonal correction factors which are different from national factors (Burke and Murphy, 2011; Denman et al., 2007). Different seasonal variations are sometimes observed in building sites on different bedrocks (Moreno et al., 2008). In the UK, this problem has been resolved by applying temperature correction factors to indoor radon measurements because the difference between the indoor and outdoor temperature has a major impact on the rate of entry of radon into dwellings (Miles, 1998). The temperature correction factors are derived from the difference between outdoor and indoor temperature for each specific measurement period (Miles, 1998; Miles, 2001).

B. Indoor Radon Measurement Validation Scheme

Great care is required in the interpretation of radon monitoring results because the rate at which radon is released into buildings is controlled by a complex series of factors, which requires monitoring equipment to be located in the right place over a prolonged period to take account of temporal variations. Validation schemes are required to (1) ensure organizations measure radon within an acceptable degree of uncertainty; (2) determine that detectors are handled in an appropriate fashion both before and after the detectors have been with householders; and (3) ensure minimum standards in how results are interpreted and presented, which includes requiring the use of the seasonal variation factors. In the United States, the EPA operates a voluntary National Radon Proficiency Program that evaluates radon measurement companies and the test services they offer. Both the UK NRPB and the U. S. EPA recommend that testing services be purchased from certified organizations.

C. Measurement of Radon in Soil Gas

Measurement of radon in soil gas using pumped monitors is recommended as the most effective method for assessing the radon potential of underlying rocks, overburden, and soil. Instruments for the determination of soil gas radon are generally based upon either an extraction method, using a “pump monitor” device for transferring a sample of the soil gas to a detector, or simply emplacing the detector in the ground (passive methods). In the former method, a thin rigid tapered hollow tube is usually hammered into the ground to a convenient depth, which causes minimum disturbance to the soil profile. Detection of radon is usually based upon the zinc sulfide scintillation method or the ionization chamber. Alpha particles produce pulses of light when they interact with zinc sulfide coated on the inside of a plastic or metal cup or a glass flask (Lucas cell). These may be counted using a photomultiplier and suitable counting circuitry. Because the radon isotopes are the only alpha-emitting gases, their concentration may be determined accurately using relatively simple equipment. Because of the different half-lives of these isotopes and their immediate daughter products, it is possible to calculate the activities of radon and thoron. The equipment is relatively robust for field use and is designed for rapid changing of the cell when it becomes contaminated. The large number of instruments produced attests to its suitability for field use. The concentration of radon in soil gases is usually sufficient that the level may be determined relatively fast; a matter of a few minutes generally suffices.

Radon can also be measured by emplacing alpha track detectors in the ground. Holes may be dug with an auger or drill to a depth of at least 0.5m and preferably 1.0m. Holes are normally lined with plastic piping in which the detector is emplaced and the top of the pipe sealed. The detectors are normally taped to the bottom of a plastic cup, which is inverted before burial. The detector is then recovered 3–4 weeks later. This procedure is used when long-term monitoring is required to overcome problems of short-term variation in radon concentration.

Although alpha track detectors overcome many of the problems associated with temporal variation in radon fluxes, they are time-consuming to emplace, requiring two visits to each site, with all the problems in reoccupying the site. More important, they require a laboratory processing stage. In practice they are generally not favored for primary investigations, although they do have an important role to play at later stages. They are also sensitive to thoron, but the presence of a polyethylene film seal, at a distance of about 5cm from the detector, reduces the amount of short half-lived thoron while having a negligible effect on radon. The polyethylene film allows radon but not water vapor to diffuse. Water droplets on the surface of the film may also affect the recorded alpha counts and water vapor absorbers may need to be introduced into the sampling device.

Papastefanou (2007) (Papastefanou, 2007) reviews the instrumentation available for measuring radon in soil gas and groundwaters.

Determination of radon potential from soil gas radon concentrations generally produces better results when soil permeability is also measured (Barnet et al., 2008; Kemski et al., 2001).

D. Measurement of Radon in Water

Radon has a high partition coefficient (gas to water) so that the passage of fine gas bubbles through water provides an efficient means of extraction. The gas may be drawn into an evacuated Lucas cell. Alternatively a sealed re-circulating system may be set up. Very careful attention must be paid to the timing of both degassing and

counting and careful calibration of the procedure with standardized radon solutions is required. Other methods require expensive equipment and laboratory processing. For example, the radon daughter ^{214}Bi photopeak emission can be measured using either a sodium iodide scintillation crystal or a high-resolution, lithium-drifted germanium semiconductor detector.

E. Measurement of Radon in Solid Materials

One of the solid daughter products of radon is ^{214}Bi . This emits high-energy gamma radiation at 1.76MeV. Gamma spectrometric determinations of uranium in the field and laboratory often make use of this photopeak on the assumption that the decay chain is in equilibrium and therefore this measurement provides an effective total radon determination. If the parent uranium mineral is resistant to weathering (e.g., thorium- and REE-rich uranium oxides and silicates, monazites, zircons, etc.) then the radium will tend to be in secular equilibrium with the uranium. In such minerals the radon loss is normally low and gamma spectrometric measurements give a good indication of the uranium contents. The measurement of radon release from solid samples requires an alternative method. Radon release from disaggregated samples (soils, stream sediments, and unconsolidated aquifer sands) may be determined by agitating a slurry of the material with distilled water in a sealed glass container, allowing a period of about 20–30 days for the generation of radon from radium, and then measuring the radon in the aqueous phase using a liquid scintillation counter. Emanation of radon from solid rock samples can be determined using a similar method.

VIII. Factors Controlling Release and Transfer of Radon Gas

Most radon remains in rocks and soil and only some of that near a free surface is released. Soil generally releases more radon than rock, as its constituents are more comminuted. The rate of release of radon from rocks and soils is largely controlled by the uranium concentration and by the types of minerals in which the uranium occurs. Once radon gas is released from minerals, the most important factors controlling its migration and accumulation in buildings include (1) transmission characteristics of the bedrock including porosity and permeability; (2) the nature of the carrier fluids, including carbon dioxide gas, surface water, and groundwater; (3) weather; (4) soil characteristics including permeability; (5) house construction characteristics; and (6) lifestyle of house occupants.

A. Mineralogical Effects

The main mineralogical factors affecting the release of radon are the solubility, internal structure, and specific surface area of uranium-bearing minerals. Uranium is very seldom homogeneously distributed throughout rocks and soils. Most of the uranium in rocks can be attributed to discrete uranium-bearing minerals, even when there is only a few mgkg^{-1} of uranium present. Because radon is a gas with a limited half-life, its chances of escaping from the parent mineral are much greater if it is generated from grain margins. Other important controls are the openness of and imperfections in the internal structure of the mineral and the specific surface area of the mineral grains.

The release of radon is generally controlled by alpha particle recoil mechanisms, which tend to expel radon from radium derived from uranium-bearing minerals. Most of the radon remains within the mineral to decay again to solid products. Only a very small proportion of the radon generated can be released by recoil. The location of the radium atoms in the mineral grains and the direction of the recoil of the radon atoms will determine whether the newly formed radon atoms enter pore spaces between mineral grains. Factors such as the specific surface area, the shape, degree of fracturing, imperfections, and even radiation-induced damage of the host uranium-bearing mineral affect the efficiency of radon expulsion. Because uranium minerals have high densities, the recoil

range is usually low. However, if radium is present in intergranular films then recoil ranges varying between 20 and 70 μm would occur. The fraction of radon, produced by radium decay, that escapes from rock or soil (called the emanation coefficient) is dependent on the surface area of the source material. Emanation coefficients are greater for rocks than minerals, whereas soils usually have the highest values. If water is present in the pore space, however, the moving radon atom slows very quickly and is more likely to stay in the pore space. However, the emanation power of soil rises with water content, probably because radon atoms ejected from soil grains have a higher probability of remaining in the pore space and migrating out of the sample if the pore space is filled with water; being twice as high as for dry soil when soil moisture is 10% (Bossew, 2003). Chau et al., (2005) suggested that the radon mass exhalation rate reaches a maximum at about 6% by weight, after which it decreases with both increasing and decreasing water content of soil samples (Chau et al., 2005). The radon emanation fraction is reported to increase markedly with grain size and reaches a constant value of 50% when moisture content is 0% and the radium is uniformly distributed on the grain surface (Sakoda et al., 2010). A dramatic increase in radon emanation occurs at smaller grain sizes with increasing moisture content because radon emanation depends greatly on the pore size between a Ra-bearing grain and a neighboring grain (Sakoda et al., 2010). Differences in the uranium-bearing minerals, and especially in the solubility of the major uranium-bearing minerals, control the amount of radon released. In some granites, for example, much of the uranium is found in the mineral uraninite (uranium oxide), which is easily weathered, especially near the surface. Uranium is more soluble in water so it is removed from the original mineral site, but the relatively insoluble radium, which is the immediate parent of radon gas, remains in a mixture of iron oxides and clay minerals. This material is a highly efficient radon generator because of the high specific surface area of the radium-bearing phase (Åkerblom and Lindgren, 1997). Radon generated from the radium can escape into the fluid phase with high efficiency thus facilitating its rapid migration to the surface. An inert gas, radon is relatively unaffected by chemical buffering reactions that often control the generation of other gases in rocks and their weathering products. In contrast, uranium in other granites may occur in chemically resistant high-thorium uraninite, zircon, monazite, and apatite, all of which liberate less radon. The mineral associations typically found in sedimentary rocks differ significantly from those in granites. In carboniferous limestone of northern England, for example, uranium is relatively uniformly distributed and associated with finely divided organic matter in the matrix of bioclastic limestones (usually $<10\text{mgkg}^{-1}\text{U}$), although it may also be concentrated in stylolites, which typically contain $20\text{--}60\text{mgkg}^{-1}\text{U}$. Even though the overall concentration of U in the limestones is below 2mgkg^{-1} , high radon emissions are probably derived from radium deposited on the surfaces of fractures and cavities. The high specific surface area of the radium permits efficient release of radon and high migration rates are promoted by the high permeability of the limestone. In addition, uranium and radium are concentrated in residual soil overlying limestone. Radium is sometimes preferentially concentrated in soil organic material, which has a high emanation coefficient (Greeman & Rose, 1996). In black shales in the UK, uranium is located mainly in the fine-grained mud matrix, where it may be present at levels up to $20\text{mgkg}^{-1}\text{U}$, and also in organic-rich bands at concentrations up to $40\text{mgkg}^{-1}\text{U}$. Much higher uranium concentrations have been reported from the Chattanooga shale in the United States ($20\text{--}80\text{mgkg}^{-1}$), the Dictyonema shale in Estonia ($30\text{--}300\text{mgkg}^{-1}$), and the alum shale in Sweden and Norway ($50\text{--}400\text{mgkg}^{-1}$). Uranium is rare in detrital phases and may also be remobilized and adsorbed on iron oxides. In sandstones, uranium is concentrated in primary detrital minerals, such as apatite and zircon, which can contain high concentrations of U ($>100\text{mgkg}^{-1}$). Uranium may also be adsorbed onto Fe oxides in the matrix of sandstone or its weathering products. Emission of radon from sandstones is restricted by the relatively low specific surface area of the uranium minerals and appears to be more dependent upon fracturing of the rock.

B. Transmission Characteristics of Bedrock

Although the generation of high levels of radon is ultimately dependent upon the concentration of uranium and upon the nature of the parent mineral, the transmission of radon gas to the surface is largely independent of these characteristics (Åkerblom & Mellander, 1997). Once the radon is released from the parent mineral into the space between mineral grains (the intergranular region) other factors take over. The most notable of these are (1) the fluid transmission characteristics of the rock including permeability, porosity, pore size distribution, and the nature of any fractures and disaggregation features and (2) the degree of water retention (saturation) of the

rocks. Faults and other fractures permit the efficient transmission of radon gas to the surface. The presence of faults with their enhanced fluid flow frequently results in high radon in soil gases (Ball et al., 1991).

C. Carrier Fluids

Radon readily diffuses into pores and cavities from mineral surfaces. However, its relatively short half-life (3.8 days) limits the distance over which diffusion may occur. In highly permeable dry gravel, radon has decayed to 10% of its original concentration over a diffusion length of 5m (UNSCEAR, 2009). In more normal soils, which are generally moist, this distance would be substantially less. Diffusive ^{222}Rn in soil gas can be determined from the specific ^{226}Ra activity, specific density, effective porosity, and radon emanation coefficients of soils and rocks (Washington & Rose, 1992). In caves, radon concentrations of approximately 100Bqm^{-3} would be expected if radon were generated by diffusion from solid limestone with $2.2\text{mgkg}^{-1}\text{U}$. However, the enhanced concentration of radon in caves suggests that structurally controlled convective transport of radon in fluids along faults, shear zones, caverns, or fractures is more significant than diffusive transport. Transportation of radon in this way may exceed 100m.

Following radon release, migration in carrier fluids, such as carbon dioxide gas or water, is considered to be the dominant means of gas transmission to the surface. Radon release is higher in rocks that have a high surface area in contact with groundwater. Once released from the uranium minerals, radon migration is dependent upon the fluid flow characteristics of the rock and soil. In water, convective or pressure gravity flow mechanisms can influence migration of the radon, whereas in a gas the transport may be controlled by the diffusion characteristics of the carrier gas. Water flow below the water table is generally relatively slow as is ground-water transport in the soil aquifer (<1–10cm per day). Thus all hydraulically transported radon will have decayed over a distance of less than 1–2m. Radon is likely to be carried away more quickly by fluids in areas of permeable rocks such as limestones. The carrier effect may also be important for other rock types. Carbon dioxide may collect radon gas in the unsaturated zone and transport it along fractures, fissures, and faults. In situations where the carbon dioxide flux is high, such as in active volcanic areas, radon may be either diluted or enhanced because of rapid transport from the generation zone to the surface.

Radon in surface water is not generally accompanied by dissolved radium. In surface stream waters, the radon concentration appears to be more closely related to the radium concentration of the stream sediment. However, the radon concentration in surface streams is usually far too low for more than a very small degree of transfer across the air/water interface to occur, unless a gas phase is introduced. Radon dissolved in subsurface fluids migrates over long distances along fractures and caverns depending on the velocity of fluid flow. Radon is soluble in water and may thus be transported for distances of up to 5km in streams flowing underground in limestone. Radon remains in solution in the water until a gas phase is introduced, for example, by turbulence or by pressure release. If emitted directly into the gas phase, as may happen above the water table, the presence of a carrier gas, such as carbon dioxide, would tend to induce migration of the radon. This appears to be the case in certain limestone formations, where underground caves and fissures enable the rapid transfer of the gas phase.

D. Weather

The principal climatic factors affecting radon concentrations are barometric pressure, rainfall, and wind velocity. In the absence of a less permeable humic or clay rich topsoil, radon concentration in soil gas varies directly with barometric pressure, and to a lesser extent, inversely with wind speed. Where the topsoil is finer grained and more humic, the effects of barometric pressure and wind velocity are reduced to a marginal role. The extent to which rainfall affects radon concentrations depends on the permeability of the soil. For permeable soils, radon concentrations are only affected during precipitation when saturation of small pore spaces with moisture effectively prohibits the rapid outgassing of radon from the soil. This causes the buildup of radon below the moisture-

saturated surface layer and increases of an order of magnitude are sometimes observed. Prolonged rainfall may penetrate deeply and seal the pore spaces in the soils to a considerable depth.

A similar buildup of radon is often observed during the night when dew forms on the surface and this can result in a twofold increase in soil gas alpha activity. Sealing of the pore spaces by near-surface moisture can result in temporary entrapment of radon in soil gases, with a significant increase in total gamma activity from radon daughters (^{214}Bi). Dry conditions cause clay-rich soils to dry out and to fracture, allowing easier egress for the soil gases and hence an increase in radon activity at the soil surface. Seasonal variation in soil pore radon concentration was observed by Rose et al. (1990) who found that the radon concentration tended to be lower in the winter and higher in the summer, often varying by a factor of 3–10. The variation was attributed largely to changes in the soil moisture content with more radon held in solution in the soil pore water during the winter. The variation is greater in the soil above 70cm depth than below this depth, presumably due to greater short-term fluctuations in soil moisture content. This suggests that radon in soil gas measurements should be taken at depths greater than 70cm in order to reduce the effects of temporal variations caused by rainfall.

Although barometric pressure and rainfall obviously cause temporal variation in radon concentration (indicated by alpha activity), it is encouraging to note that the soil gas radon fluxes in areas that are not mineralized appear to be relatively uniform. Various rock types have been tested and the site variation is often less than that between adjacent rock types. This is particularly important for radon potential mapping based on the measurement of radon in soil gas.

E. Soil Characteristics

The principal soil properties that influence the concentration of radon in soil gas, including the rate of release of radon and its transfer through soils, are soil permeability and soil moisture. In general soil permeability depends on such factors as soil texture, structure, median pore diameter, pore size distribution, pore volume, packing density, soil bulk density, and grain size. Soil mineralogy is an important factor controlling soil gas radon concentrations; in some cases, organically bound ^{226}Ra can be a principal source of ^{222}Rn in soil gas (Greeman et al., 1990; Greeman & Rose, 1996). Radon volume activity increases with the percentage of coarse material in the soil thus confirming the general correlation between radon fluxes and soil permeability. In general, coarse gravelly soils will tend to have higher radon fluxes than impermeable clay soils. However, humic and clay soils may be impermeable in the winter when saturated with water or filled with ice if the ground is frozen, and during very dry periods they may crack and behave in a permeable manner. Soil permeability and rainfall (soil saturation) exert a considerable control on radon concentrations in houses. Soil permeability generally closely reflects the permeability of the underlying rocks and superficial deposits such as glacial till, alluvium, or gravel. Radon diffuses more slowly through water than air, so water-saturated soils impede the diffusion of radon enough for it to decay to harmless levels before it has diffused more than 5–10cm. Consequently, radon from water-saturated soils is unlikely to enter buildings unless it is transported in other gases such as carbon dioxide or methane.

It is important to remember that whereas the top meter of the soil profile is generally removed during the construction of foundations for a dwelling, only a few centimeters of topsoil are removed from the remainder of the subfloor space. Indeed, in many cases the soil profile beneath a dwelling will not be unduly influenced by temporal variations in rainfall. The influence of the geochemistry and permeability of the bedrock or overburden beneath a dwelling on the potential for radon emissions from the ground may be greater than near-surface soil properties.

Only 10–50% of the radon produced in most soils escapes from the mineral grains and enters the pores. Soils in the United States generally contain between 5 and 80BqL^{-1} of radon. Drier, highly permeable soils and bedrock—such as limestones, coarse glacial deposits, and fractured or cavernous bedrock, and hill slopes—are usually associated with relatively high levels of indoor radon. The permeability of the ground permits radon-bearing air to move greater distances before it decays, and thus contributes to high indoor radon even if the radon content of soil gas is in the normal range ($5\text{--}50\text{BqL}^{-1}$).

IX. Radon Migration Pathways

When considering *natural migration pathways*, it should be noted that although the general direction and position of planar discontinuities and openings including bedding planes, joints, shear zones, and faults can be determined by detailed structural mapping, the precise location of such migration pathways is often difficult to establish, especially if the area is covered with soil or drift. In the United States, high radon is associated with U-enriched shear zones in granites, which are characterized by high radon in soil gas and groundwater. Indeed, some of the highest indoor radon levels in the United States are associated with sheared fault zones. Similar observations have been made in southwest England. Radon and other gases are known to concentrate and migrate upward along faults and through caves and other solution cavities. However, natural cavities such as potholes and swallow holes in limestone would also be difficult to locate precisely due to their irregular and relatively unpredictable disposition.

Radon has been used to identify the location of faults and frequently reaches a maximum in the direct vicinity of faults (Barnet et al., 2008; Ielsch et al., 2010; Neri et al., 2011; Pereira et al., 2010; Swakon et al., 2005). However, a consistent decrease of indoor radon away from mapped faults is not observed in the UK (Appleton, 2004) and it is likely that elevated radon concentrations will be associated mainly with active faults. Radon has also been used for earthquake and volcanological investigations (Avino et al., 1999; Ciotoli et al., 1998; Ghosh et al., 2009; Hishinuma et al., 1999; Papastefanou, 2010)

Artificial pathways underground include mine workings and disused tunnels and shafts. Radon concentrations in old uranium and other mine workings are commonly 10,000–60,000 Bq m⁻³ and can be as high as 7,100,000 Bq m⁻³ even when uranium is a very minor component of the metalliferous veins (Gilmore et al., 2001). High radon is known to be associated with gassy ground overlying coal-bearing rock strata. In addition, relatively randomly orientated and distributed blasting and subsidence fractures will affect areas underlain by mined strata. The sites and disposition of recent coal mine workings in some countries may be obtained from mine records, although these may not be reliable. Other artificial pathways related to near-surface installations include electricity, gas, water, sewage, and telecommunications services, the location of which may be obtained from the local service agencies. Land drains provide another potential migration pathway. The detection and prediction of migration pathways is difficult and may be imprecise, although a detailed geological and historical assessment together with appropriate radon gas monitoring and a detailed site investigation should provide a reasonable assessment of the source and radon gas migration pathways. Information on the local geology may be obtained from maps, memoirs, boreholes, and site investigation records.

X. Factors Affecting

Radon in Buildings

The design, construction, and ventilation of the home affect indoor radon levels. Radon can enter a home through cracks in solid floors and walls below construction level; through gaps in suspended concrete and timber floors and around service pipes; and through crawl spaces, cavities in walls, construction joints, and small cracks or pores in hollow-block walls (Figure 3). Radon concentrations are generally highest in basements and ground floor rooms that are in contact with the soil or bedrock. Air released by well water during showering and other household activities may also contribute to indoor radon levels, although this generally makes a relatively small contribution to the total radon level.

When constructing a house with a basement in the United States, a hole is dug, footings are set, and coarse gravel is usually laid down as a base for the basement slab. The gap between the basement walls and the ground outside is backfilled with material that often is more permeable than the original ground. Radon moves into this permeable material and the gravel bed underneath the slab from the surrounding soil. The backfill material is typically rocks and soil from the foundation site but may be imported material with different radon emanation characteristics to the local rocks and soils. Therefore, the amount of radon in the permeable material depends on

the amount of uranium in the local or imported rock as well as the type, permeability, and moisture content of the soil. The backfill layer will need to have a thickness of at least 1m or have a very high Ra²²⁶ concentration for it to be a significant source of radon in the building.

In a typical masonry building in which radon occurs at the UK national average level of 20Bqm⁻³, approximately 60% of radon comes from the ground on which the building stands, 25% from building materials, 12% from fresh air, 2% from the water supply, and 1% from the gas supply. These figures apply to the average house in the UK, but can vary substantially, and the proportion of radon entering a home from the ground will normally be much higher in homes with high radon levels. The dominant mechanism of radon ingress is pressure-induced flow through cracks and holes in the floor. Slightly negative pressure differences between indoor and outdoor atmospheres caused by wind outside and heating inside the building draw radon contaminated air into the building, especially through the floor. Energy-conserving measures such as double-glazing restrict the fresh supply of air and lessen the dilution of radon indoors. Conversely, they may also reduce the pressure difference between indoors and outdoors and thus reduce the influx of radon from the ground. Poor ventilation may increase radon concentrations, but it is not the fundamental cause of high indoor radon levels. Household energy efficiency improvements that decrease ventilation (e.g. better sealed windows and doors) could lead to an increase in exposure to radon.

Indoor radon concentrations are generally about 1000 times lower than radon in the soil underlying the house. Most houses draw less than one percent of their indoor air from the soil with the remainder from outdoors where the air is generally quite low in radon. In contrast, houses with low indoor air pressures, poorly sealed foundations, and several entry points for soil air may draw as much as 20% of their indoor air from the soil. Consequently, radon levels inside the house may be very high even in situations where the soil air has only moderate amounts of radon.

Clavensjö and Åkerblom (1994) suggested that the ²²²Rn concentration in a building or room that results from the transport of soil air may be calculated by the following formula:

$$C_{\text{building}} = \frac{C_t \cdot L}{V_{\text{building}} \cdot (n + \lambda)}$$

C_{building}	= ²²² Rn concentration in the building/room (Bq m ⁻³)
C_t	= ²²² Rn concentration in the soil air entering from the ground (Bq m ⁻³)
L	= volume soil air entering from the ground (m ³ h ⁻¹)
n	= rate of air change in the building (air changes h ⁻¹)
λ	= decay constant, for ²²² Rn, 7.55 · 10 ⁻³ (h ⁻¹)
V_{building}	= building/room volume (m ³)

In high-rise residential buildings without basements in which most people are living in apartments, the contribution to indoor radon from the ground may be expected to be insignificant. In Hong Kong, where the geometric mean indoor radon concentration for residential buildings is quite high (48 Bq m⁻³), the relatively small contribution from soil gas may be redistributed through the high-rise residential buildings by centralized air conditioning systems (Mui et al., 2008), although the main source of indoor radon in Hong Kong is likely to be concrete containing granite aggregate.

XI. Radon Potential Mapping Methods

Accurate mapping of radon-prone areas helps to ensure that the health of occupants of new and existing dwellings and workplaces is adequately protected. Radon potential maps have important applications, particularly in the control of radon through planning, building control, and environmental health legislation. Radon potential

maps can be used (1) to assess whether radon protective (preventive) measures may be required in new buildings; (2) for the cost-effective targeting of radon monitoring in existing dwellings and workplaces to detect those buildings with radon above reference levels which need to be remediated; (3) to allow measurement campaigns and public awareness to be targeted on areas at greatest risk and (4) to provide radon risk assessments for homebuyers and sellers. The radon map data helps local authorities to communicate quickly and effectively with existing homeowners or developers planning to build new homes in high radon areas. In the UK, this is achieved by the online reports services developed by BGS and the HPA: (i) for existing homes with a valid postcode, a Radon Risk Report can be ordered online from the UK Radon website (<http://www.ukradon.org/>); (ii) for existing large homes and other large buildings and plots of land, a GeoReport can be obtained from BGS (<http://shop.bgs.ac.uk/georeports/>). It is important, however, to realize that radon levels often vary widely between adjacent buildings due to differences in the radon potential of the underlying ground as well as differences in construction style and use. Whereas a radon potential map can indicate the relative radon risk for a building in a particular locality, it cannot predict the radon risk for an individual building. In the UK, radon potential maps generally indicate the probability that new or existing houses will exceed a radon reference level, which in the UK is called the action level (200Bqm³) (Miles and Appleton, 2005; Miles et al., 2007). In other countries, geological radon potential maps predict the average indoor radon concentration (United States) or give a more qualitative indication of radon risk (Germany and the Czech Republic).

Two main procedures have been used for mapping radon-prone areas. The first uses radon measurements in existing dwellings to map the variation of radon potential between administrative or postal districts or grid squares. The second is geological radon potential mapping in which each geological feature is assigned to a radon potential class based on the interpretation of one or more of the following types of data: (1) radon concentrations in dwellings (indoor radon), (2) concentration, mineralogical occurrence, and chemical state of uranium and radium in the ground (radiometric and geochemical data), (3) rock and soil permeability and moisture content, (4) concentration of radon in soil gas, and (5) building architecture (construction characteristics). Because the purpose of maps of radon-prone areas is to indicate radon levels in buildings, maps based on actual measurements of radon in buildings are generally preferable to those based on other data.

Procedures for monitoring and surveys of radon in dwellings are described in Nazaroff (1988) and Miles (2001). In the UK, measurements are made with passive integrating detectors over a period of three months whereas short-term "screening" measurements taken over a 2- to 7-day period are commonly used for mapping in the United States. Measurements carried out over less than a year should be corrected for seasonal variations. In the United States, houses with basements typically have higher indoor radon than those with slab-on-grade construction because basements tend to have more entry points for radon and a lower internal pressure relative to the soil than non-basement homes. Architecture type is one factor within the Radon Index Matrix used to estimate geological radon potential in the United States (Gundersen & Schumann, 1996).

Bungalows and detached houses tend to have higher indoor radon than terraced houses or flats in the same area of the UK. Building material, double-glazing, draught-proofing, date of building, and ownership also have a significant impact on indoor radon concentrations. Radon potential mapping is sometimes based on indoor radon data that have been normalized to a mix of houses typical of the housing stock as this removes possible distortion caused by construction characteristics. Maps based on results corrected for temperature but not normalized to a standard house mix reflect such factors as the greater prevalence of detached dwellings in rural areas, and hence the higher risk of high radon levels in rural areas compared with cities where flats are usually more prevalent. Radon potential estimates based on radon levels in the actual housing stock are more appropriate for the identification of existing dwellings with high radon.

Requirements for mapping radon-prone areas using indoor radon data are similar whether the maps are made on the basis of grid squares or geological units. These requirements include (1) accurate radon measurements made using a reliable and consistent protocol, (2) centralized data holdings, (3) sufficient data evenly spread, and (4) automatic conversion of addresses to geographical coordinates. It appears that Great Britain is the only country that currently meets all of these requirements for large areas (Miles and Appleton, 2005). In countries where lesser quality or quantity of indoor radon data are available, there is greater reliance on proxy data for radon potential mapping (e.g., Czech Republic, Germany, Sweden, and United States). Where there are no existing houses or indoor radon measurements, proxy data (such as soil gas radon concentrations) are required to map radon potential.

Mapping levels of radon in administrative areas has the advantage of simplifying any subsequent administrative action. Use of grid squares allows an appropriate size of area to be chosen and simplifies the analysis. Use of geological boundaries may help to delineate differences in radon potential with greater spatial accuracy than other types of boundary. Whereas a wide variety of factors affect the concentration of radon in buildings, regional variations are related principally to the geological characteristics of the ground. Indoor radon surveys in the UK have confirmed the association of high levels of radon in dwellings with uraniferous granites, uraniferous sedimentary rocks, permeable limestones, and phosphatic ironstones, as well as fault and shear zones. Similar observations have been made in the Czech Republic, Germany, Luxembourg, Sweden, and the United States.

It is important to remember that however indoor radon data are grouped (whether by grid square or geological unit), a wide range of indoor radon levels is likely to be found. This is because there is a long chain of factors that influence the radon level found in a building, such as radium content and permeability of the ground below it, and construction details of the building (Miles & Appleton, 2000). Radon potential does not indicate whether a building constructed on a particular site will have a radon concentration that exceeds a reference level. This can only be established through measuring radon in the building.

Uncertainties relating to house-specific factors, proximity to geological boundaries and measurement error impact on the radon mapping process (Hunter et al., 2009, 2011)(Bossew, 2009).

A. Non-Geological Radon Potential Mapping

Radon measurements in existing dwellings are used to map the radon potential of countries (Figure 4), administrative districts, or grid squares without taking into consideration the geological controls on radon in dwellings. Because the factors that influence radon concentrations in buildings are largely independent and multiplicative, the distribution of radon concentrations is usually lognormal, so lognormal modeling can be used to produce accurate estimates of the proportion of homes above a reference level (Miles, 1998)(Bossew, 2010), although other transformations have also been investigated (Murphy and Organo, 2008). In the UK, radon maps were used to show the fraction of the housing stock above the action level in each 5-km grid square (Figure 5) (Lomas et al., 1996). Where house radon data are plentiful, maps using grid squares smaller than 5km can be made. In some cases, this method can show up variations that are obscured by general geological grouping, such as variations in radon potential within a geological unit. Investigations in southwestern England revealed that the finer the grid, the closer the correlation with the geological controls of radon in dwellings.

Radon potential mapping using indoor radon measurements has been carried out in other European countries including Ireland, Luxembourg and France, (Dubois, 2005; Fennell et al., 2002; Organo and Murphy, 2007) but the maps are not as detailed as the 5-km grid maps of the UK. This is mainly due to the relatively low measurement density and restricted coverage. In India and China, for example, average indoor radon concentrations of approximately 23 Bq m^{-3} in both countries are derived from national surveys of only 1500 and 11,000 dwellings respectively ((Cheng et al., 2002; Jin et al., 1996; Ramachandan and Sathish, 2011) compared with over 4,500,000 available for radon mapping in the UK.

In the United States most measurements of indoor radon have been made using short-term charcoal monitors, so these cannot be used directly to estimate long-term average radon levels. Although individual short-term measurement results are poor indicators of radon potential, aggregations of them can be corrected for bias and can provide useful information where no long-term results are available. A statistical technique known as Bayesian analysis improves estimates of mean radon level in areas where the data are sparse. A U. S. EPA survey of radon in homes covered about 6000 homes across the United States, all measured using long-term etched track detectors. The data distribution is very sparse, given the size of the country, but it can provide estimates of the mean radon levels and distributions for the whole country and states, although not for smaller areas.

A range of statistical and geostatistical methods have been employed in some countries to map the spatial variation in indoor radon concentrations ((Bossew, 2009; Bossew et al., 2008; Dubois et al., 2007; Dubois et al., 2010; Friedmann and Gröller, 2010; Raspa et al., 2010; Tuia and Kanevski, 2008; Zhu et al., 2001).

B. Geological Radon Potential Mapping

Approximately 25% of the total variation of indoor radon concentrations in England and Wales can be explained by the mapped bedrock and superficial geology. The proportion of the total variation explained by geology is higher (up to 37%) in areas where there is strong contrast between the radon potential of sedimentary geological units and lower (14%) where the influence of confounding geological controls, such as uranium mineralisation, cut across mapped geological boundaries. The proportion of the variation that can be attributed to mapped geological units increases with the level of detail of the digital geological data (Appleton and Miles, 2010). Bossew et al. (2008) showed that the fraction of the variation in indoor radon concentrations in Austria explained by geology was 11.2% when the approximately 24 000 short-term indoor radon measurements were grouped into 111 relatively detailed geological units. As a consequence, the most accurate and detailed radon potential maps are generally those based on house radon data and geological boundaries provided that the indoor radon data can be grouped by sufficiently accurate geological boundaries. In the absence of an adequate number of high quality indoor radon measurements, proxy indicators such as soil gas radon data or information on U content may be used to assess geological radon potential. The reliability of maps based on proxy data increases with the number of classes as well as the quantity and quality of available data. Radon potential maps based on indoor radon data grouped by geological unit have the capacity to accurately estimate the percentage of dwellings affected together with the spatial detail and precision conferred by the geological map data (Miles & Ball 1996). The reliability and spatial precision of mapping methods is, in general, proportional to the indoor radon measurement density. It is, however, reassuring that even when the measurement density is as low as the minimum for 5-km grid square mapping (i.e., 0.2–0.4perkm²), geological radon potential mapping discriminates between geological units in a logical way. These relationships can be explained on the basis of the petrology, chemistry, and permeability of the rock units and are confirmed in adjoining map sheets with higher measurement densities (Miles & Appleton, 2000).

Geological radon potential maps of the UK have been produced at 1:625,000, 1:250,000, and 1:50,000. Each geological unit within a map sheet or smaller area, such as a 5-km grid square, has a characteristic geological radon potential that is frequently very different from the average radon potential for the grid square shown (Figure 6). Lithological variations within geological units can cause geological radon potential mapping to miss significant areas of higher radon potential identified by 1-km grid square mapping. Geological and grid square mapping are likely to be most powerful when used in a complementary fashion by integrating maps produced by the two methods and by grouping results both by geological unit and by grid square (Appleton & Miles, 2002).

The factors that influence radon concentrations in buildings are largely independent and multiplicative, the distribution of radon concentrations is usually lognormal. Therefore lognormal modelling was used to produce accurate estimates of the proportion of homes above a reference level in the UK (Miles et al., 2007). Radon maps based on indoor radon data grouped by geological unit have the capacity to accurately estimate the percentage of dwellings affected together with the spatial detail and precision conferred by the geological map data. When indoor radon measurements are grouped by geology and 1-km squares of the UK national grid, the cumulative percentage of the variation between and within mapped geological units is shown to be 34–40% (Appleton and Miles, 2010). This confirms the importance of radon maps that show the variation of indoor radon concentrations both between and within mapped geological boundaries. Combining the grid square and geological mapping methods gives more accurate maps than either method can provide separately (Figure 7) (Miles and Appleton, 2005).

In Austria, indoor radon data, normalised to a defined 'standard room' to reduce variations related to house type, are combined with geological information to produce a national radon potential map (Friedmann and Gröller, 2009).

A combination of indoor radon measurements and geological information are used to produce radon potential maps in Canada (Chen, 2009; Chen et al., 2011), Radon potential maps for highly populated areas are produced using indoor radon measurements where enough indoor radon data are available, whilst in areas where few or no indoor radon measurements are available, radon potential maps could be developed from various data sources with a multi-factor scoring system including geological information on soil permeability, soil gas radon concentra-

tion and ground uranium concentration, although both these approaches are constrained by the sparsity of indoor radon measurements and geological information relevant to radon (Chen, 2009).

Uranium and radium concentrations in surface rocks and soils are useful indicators of the potential for radon emissions from the ground. Uranium can be estimated by gamma spectrometry either in the laboratory or by ground, vehicle, or airborne surveys (IAEA, 2003; Dickson, 2004). The close correlation between airborne radiometric measurements and indoor radon concentrations has been demonstrated in the United States in Virginia and New Jersey, Nova Scotia in Canada, and also in parts of the UK (Appleton & Ball, 2001; Appleton et al., 2008, 2011; Scheib et al., 2006), Ireland (Appleton et al., 2011), Italy (Verdoya et al., 2009), Norway (Smethurst et al., 2008) and Thailand (Wattanakorn et al., 2007). Areas with high permeability tend to have significantly higher indoor radon levels than would be otherwise expected from the equivalent ^{226}Ra concentrations, reflecting an enhanced radon flux from permeable ground.

Duval and Otton (1990) identified a linear relationship between average indoor radon levels and surface radium content for soils of low to moderate permeability. However, areas with high permeability ($>50\text{cmh}^{-1}$) had significantly higher indoor radon levels than would otherwise be expected from the ^{226}Ra concentrations, which reflects an enhanced radon flux from permeable ground. Grasty (1997) demonstrated that any estimate of natural gamma ray flux from the uranium decay series (i.e., radium) in the ground must take into consideration the radon coefficient of the soil as well as its radon diffusion coefficient, which depends largely on soil moisture. Clay soils tend to have higher eU when wet whereas sandy soils have lower eU (Grasty, 1997).

Sweden was the first country to make use of airborne gamma ray spectrometry data to produce maps of radon potential. Radon potential is estimated and mapped on the basis of available data including (1) geology, (2) airborne radiometric surveys (covering 65% of Sweden), (3) results from radiometric surveys of the ground, (4) results from radon surveys in buildings, (5) results from earlier geotechnical investigations (e.g., permeability and groundwater level), (6) field surveys including gamma spectrometry, and (7) orientation soil gas radon measurements. Åkerblom (1987) established a simple threefold radon risk classification based on geology, permeability, and soil gas radon (Table IX). These criteria are used at a mapping scale of 1:50,000 or larger in conjunction with airborne gamma spectrometry surveys to produce provisional radon risk maps both for indoor radon and groundwater (Åkerblom and Lindgren, 1997). A similar classification was used for the radon risk map of Estonia (Petersell et al, 2005)

Radon risk mapping of the Czech Republic at a scale of 1:500,000 (Figure 8) is based upon a number of data sets for airborne radiometry, geology, pedology, hydrogeology, ground radiometry, and soil gas radon. Rock and soil permeability were obtained from hydrogeological and pedological maps and reports. Radon risk categories (low, medium, and high) were established for geological and lithological units and were based upon a rigid set of rules accepted by the Ministry of Environment. Radon risk maps are currently produced at the 1:50,000 scale, and these can be used for the identification of dwellings exceeding the guidance level to an accuracy of 70–80% (Mikšová & Barnet, 2002; Barnet et al., 2008). However, the maps are not recommended for the prediction of the requirement for radon protective measures in new buildings for which soil gas radon site assessments are required.

Airborne radiometric survey data were used to produce the first radon potential maps in the United States. The U. S. EPA radon map was developed using five factors to determine radon potential (indoor radon measurements, geology, aerial radioactivity, soil permeability, and foundation type). Radon potential assessment is based on geologic provinces adapted to county boundaries for the Map of Radon Zones (Figure 9). The purpose of the map is to assist national, state, and local organizations to implement radon-resistant building codes. In the United States, high geological radon potential is associated with granites, limestones, black shales, and glacial tills and gravels derived from these bedrocks in the Appalachians; sandy and clay tills derived from sandstones, limestones, and black shales in the northern Great Plains and Great Lakes areas; and uraniumiferous granites, permeable limestones, sedimentary, and metamorphic rocks together with derived colluvial and alluvial deposits in the Rocky Mountains and parts of the western Great Plains.

In Northern Ireland (Appleton et al., 2008; Appleton et al., 2011c) linear regression analysis of airborne and soil geochemical parameters revealed that the most significant independent variables were eU, a parameter derived from airborne gamma spectrometry measurements of radon decay products in the top layer of soil and exposed bedrock, and the permeability of the ground. The radon potential map generated from airborne gamma spectrometry data agrees in many respects with the map based on indoor radon data and geology but there are several areas where radon potential predicted from the airborne radiometric and permeability data is substantially lower. This under-prediction could be caused by the radon concentration being lower in the top 30 cm of the soil than at greater depth, because of the loss of radon from the surface rocks and soils to air.

Smethurst et al (2008) found a simple linear relationship between uranium concentration in the ground and the proportion of dwellings with radon concentrations above 200 Bq m^{-3} and demonstrated the spatial correlation between high concentrations of equivalent uranium and radon concentrations in dwellings, permitting the use of airborne radiometric data in radon hazard mapping, especially to classify areas with few indoor radon measurements. In areas of high permeable fluvio-glacial deposits, the airborne data underestimate the level of the radon hazard caused by the high ground permeability, thus confirming the observations of Scheib et al. (2006), and Appleton et al (2008, 2011a, 2011b) that in some geological domains, the permeability of the ground needs to be factored in to modeling of indoor radon from airborne radiometric data. In both Norway and Northern Ireland, additional areas of high potential radon hazard, not yet confirmed by indoor radon measurements, were identified.

The probability of homes in the Republic of Ireland having high indoor radon concentrations is currently estimated on the basis of known in-house radon measurements averaged over $10 \times 10 \text{ km}$ grid squares (Fennell et al., 2002). Airborne gamma-ray spectrometer data was used to predict radon potential in two distinct areas of Ireland using linear regression models derived from in-house radon measurements in conjunction with geological and ground permeability data. The best agreement between the percentage of dwellings exceeding the Reference Level (RL) for radon concentrations in Ireland ($\%>RL$), estimated from indoor radon data, and modelled RP in the Tralee-Castleisland area is produced using models based on airborne gamma-ray spectrometry equivalent uranium (eU) and ground permeability data. Good agreement was obtained between the $\%>RL$ from indoor radon data and radon potential (RP) estimated from eU data in the Cavan area using terrain specific models. In both areas, RP maps derived from eU data are spatially more detailed than the published 10-km grid map (Appleton et al., 2011b).

A harmonised methodology to derive a single map of the geogenic radon potential is being developed. In order to improve regulatory tools for radon risk management in France, the method aims to determine the capacity of geological units to produce radon and facilitate its transfer, based on the interpretation of existing geological data. Geological units are classified according to their uranium content in order to create a radon source potential map. This initial map is enhanced by overlaying parameters, such as fault lines, which control the preferential pathways of radon through the ground and which can increase the radon levels in soils. The geogenic radon map provides a more precise zoning than the scale of the map of radon priority areas currently based solely on administrative boundaries (Ielsch et al., 2010).

After uranium and radium concentration, the permeability and moisture content of rocks and soils is probably the next most significant factor influencing the concentration of radon in soil gas and buildings. Radon diffuses farther in air than in water, so in unsaturated rocks and overburden with high fluid permeability, higher radon values are likely to result from a given concentration of uranium and radium than in less permeable or water-saturated materials. Weathering processes can also affect permeability. Enhanced radon in soil gas is also associated with high-permeability features such as fractures, faults, and joints. The fracturing of clays, resulting in enhanced permeability, combined with their relatively high radium content and their emanation efficiency may also result in higher radon concentrations in dwellings. The permeability of glacial deposits exerts a particularly strong influence on the radon potential of underlying bedrock.

It has been demonstrated in a number of countries, including Canada, Germany, the UK, the United States, and Sweden, that soil gas radon measurements combined with an assessment of ground permeability can be used to map geological radon potential in the absence of sufficient indoor radon measurements. Significant corre-

lations between average indoor and soil gas radon concentrations, grouped according to geological unit, have been recorded in the Czech Republic (Figure 10), Germany, the UK, and the United States.

Soil gas radon measurements combined with an assessment of ground permeability has been used to map geological radon potential in the absence of sufficient indoor radon measurements. 10 to 15 soil gas radon measurements are generally required to characterize a site or geological unit. Ground classifications based on geology, permeability and soil gas radon measurement have been developed in Sweden (Clavensjö and Åkerblom, 1994), Finland, Germany (Kemski et al., 2008), Estonia and the Czech Republic (Barnet et al., 2008).

In Germany, Kemski and Klingel (2008)(Kemski and Klingel, 2008) demonstrated that soil gas and indoor radon activity concentrations are connected by a transfer factor and that this factor can vary depending on regional differences in building characteristics, especially between the western and eastern part of Germany. Risk prediction maps for radon in houses can be derived from soil gas radon map data and the regional distribution of the radon transfer factor to produce a map indicating the probability of exceeding indoor radon threshold values. The probability maps can be used for indoor radon prediction, especially in regions without a sufficient number of indoor radon data (Kemski and Klingel, 2008).

Where low correlations have been measured between radon in soil gas and radon in adjacent houses, the probable causes include: (1) the small number of houses with variable design in the study, (2) single rather than multiple soil gas measurements, (3) short-term indoor radon measurements, and (4) a mixture of summer and winter measurements. Spot measurements of soil gas radon and short-term indoor data are known to be relatively unreliable and it is now generally accepted that 10–15 soil gas radon measurements are required to characterize a site or geological unit.

XII. Radon Site Investigation Methods

Radon migrates into buildings as a trace component of soil gas. Therefore the concentration of radon in soil gas should provide a good indication of the potential risk of radon entering a building if its construction characteristics permit the entry of soil gas. There is a growing body of evidence that supports the hypothesis that soil gas radon is a relatively reliable indirect indicator of indoor radon levels at the local as well as the national scale (Figures 10 and 11). However, In some cases, soil gas radon data may be difficult to interpret due to the effects of large diurnal and seasonal variations in soil gas radon close to the ground surface as well as variations in soil gas radon on a scale of a few meters.

Soil gas radon data may be difficult to interpret due to the effects of large diurnal and seasonal variations in soil gas radon close to the ground surface and variations in soil gas radon on a scale of a few meters. The former problem may be overcome by sampling at a depth greater than 70cm or by the use of passive detectors with relatively long integrating times, although this may not be a practical option if site investigation results are required rapidly. Small-scale variability in soil gas radon may be overcome by taking 10–15 soil gas radon measurements on a 5- to 10-m grid to characterize a site. Radon in soil gas varies with climatic changes including soil moisture, temperature, and atmospheric pressure. Weather conditions should be as stable as possible during the course of a soil gas radon survey. A range of methods such as controlled gas extraction, air injection procedures, or water percolation tests can be used to estimate gas permeability at a specific site. In the absence of permeability measurements, more qualitative estimates of permeability can be based on visual examination of soil characteristics, published soil survey information, or on the relative ease with which a soil gas sample is extracted.

In some areas and under some climatic conditions, site investigations using soil gas radon cannot be carried out reliably, for example, when soil gas cannot be obtained from waterlogged soils or when soil gas radon concentrations are abnormally enhanced due to the sealing effect of soil moisture. These conditions are particularly common in winter. Problems with the determination of permeability and its incorporation into a radon site investigation procedure have been encountered in the Czech Republic where the quality of the permeability classification obtained at a site is very reliant on the personal experience of the technical staff carrying out the site investi-

gation. If soil gas radon concentrations cannot be determined because of climatic factors, measurement of radon emanation in the laboratory or gamma spectrometric measurement of eU can be used as radon potential indicators in some geological environments. However, few data are available and the methods have not been fully tested.

The Swedish National Board of Housing, Building, and Planning has adopted a ground classification based on geology, permeability, and soil gas radon measurement.(Clavensjö and Åkerblom, 1994) This procedure is used to predict radon emissions expected on a particular construction site. Finland has adopted a similar radon risk classification of building ground based on radioactivity and permeability. Measurement of radon emanation coefficients and radium concentrations by gamma spectrometry is also used to investigate radon characteristics of the ground in new building areas where buildings are to be constructed on unconsolidated sediments or directly onto bedrock (Table VIII) (Clavensjö & Åkerblom, 1994). In Germany an empirical ranking classification has been developed for radon potential based on median soil gas radon and permeability measured by air injection through the soil gas probe. All new development sites in the Czech Republic require a site investigation comprising a geological survey and measurement of radon in soil gas. The radon risk classification (Table IX) is based upon soil gas radon concentration limits and is broadly similar to classifications used in Finland, the UK, the United States, and Sweden. The Czech Radon Risk Classification For Foundation Soils (Table X) has now been replaced by a classification (Figure 12) based on the determination of the building site radon index (Barnet et al., 2008).

Radon may also be a problem underground, such as in tunnels. Radon emanation from borehole core samples can be determined to derive values of radon emanation per unit of surface area, which is an effective and simple way of assessing radon hazard in tunnels at the site investigation stage (Talbot et al., 1997).

 XIII. Geological Associations

Geology exerts a strong control on indoor radon and radon hazard. Relatively high concentrations of radon are associated with particular types of bedrock and unconsolidated deposits, for example some granites, uranium-enriched phosphatic rocks and black shales, limestones, sedimentary ironstones, permeable sandstones and uraniumiferous metamorphic rocks. Fault shear zones, glacial deposits derived from uranium-bearing rock and even clay-rich sediments from which high radon emanation reflects large surface area and high permeability caused by cracking, may be radon prone along with uranium and radium enriched soils derived from limestone (especially karstic terrain), uranium mining residues, and mine tailings.

Rock types that are high radon sources in the United States include:

1. Uraniferous metamorphic rocks and granites—some of the highest indoor levels in the United States, particularly in the Rocky and Appalachian ranges and the Sierra Nevada are associated with fault shear zones in these rocks
2. Marine black shales are sources of high radon throughout the United States and especially in the central region from Ohio to Colorado
3. Glacial deposits derived from uranium-bearing rock and sediment, especially in the northwestern Midwest, where high radon emanation reflects large surface area and high permeability caused by cracking when dry
4. Soils derived from carbonate, especially karstic terrain, which are high in uranium and radium
5. Uranium mining residues and mine tailings in the states of the western United States (e.g., Colorado)
6. Phosphate ore close to the surface and in mining waste on the surface, can result in high radon concentrations, especially in Polk County, Florida.

The maximum ^{226}Ra concentration in phosphate ores is typically about 50 times greater than the average concentration in soil. Releases from coal residues and the burning of natural gas and coal complete the list of major contributors to atmospheric radon in the USA (Gundersen et al., 1992).

Enhanced levels of radon in houses and soil gas in the UK are associated with the following geological associations. The *uranium* association comprises rocks and their weathering products containing enhanced levels of uranium or radium. The *permeable rock* association comprises permeable rocks, unconsolidated overburden, and their weathering products. Areas underlain by less permeable rocks, unconsolidated overburden, and soils, especially where these have low uranium concentrations, are generally characterized by low radon in houses and soil. Low radon is also associated with permeable sandstones containing low quantities of uranium.

The uranium association comprises granites in southwestern England characterized by high uranium concentrations, a deep weathering profile, and uranium in a mineral phase that is easily weathered. Although the uranium may be removed through weathering, radium generally remains *in situ* (Ball & Miles, 1993). Radon is easily emanated from the host rock and high values of radon have been measured in groundwaters and surface waters ($110\text{--}740\text{BqL}^{-1}$) and also in soil gas (frequently $>400\text{BqL}^{-1}$). There is a clear correspondence between areas where more than 30% of the house radon levels are above the action level and the major granite areas (Ball & Miles, 1993). In SW England, the Grampian and Helmsdale areas of Scotland (Scheib et al., 2009) and the Mourne Mountains in Northern Ireland (Appleton et al., 2011c) there is a correlation between areas where it is estimated that more than 30% of the house radon levels are above 200 Bq m^{-3} and the major granite areas. Groundwater wells with $>500\text{ B L}^{-1}$ are associated with uranium rich granite bedrock in Sweden, especially when fractures are coated with ^{226}Ra , and also with areas containing many pegmatitic intrusions (Akerblom and Lindgren, 1997)

The depositional and diagenetic environment of many black shales leads to enrichment of uranium. For example, some Lower Carboniferous shales in northern England and NE Wales contain $5\text{--}60\text{mgkg}^{-1}$ uranium. Weathering and secondary enrichment can substantially enhance U levels in soils derived from these shales. It is found that on average 15–20% of houses (rising to more than 65% in some areas) sited on uraniumiferous black shales with $>60\text{mgkg}^{-1}\text{U}$ and high soil gas radon (32BqL^{-1} ; Ball et al., 1992) are above the UK radon action level.

Uranium-enriched phosphatic horizons occur in the Carboniferous Limestone, the Jurassic oolitic limestones, and in the basal Cretaceous Chalk in the UK and these sometimes give rise to high radon in soil gases and houses. Many iron deposits are phosphatic and slightly uraniferous and a large proportion (>20%) of houses underlain by the Northampton Sand Formation (NSF) ironstone in England are affected by high levels of radon (Figures 5 and 11). Phosphatic pebbles from the Upper Jurassic, and Lower and Upper Cretaceous phosphorite horizons in England contain 30–119mgkg⁻¹U. Radon in dwellings is a significant problem in areas where these phosphatic rocks occur close to the surface, especially if the host rocks are relatively permeable. The NSF consists of ferruginous sandstones and oolitic ironstone with a basal layer up to 30cm thick containing phosphatic pebbles. Whereas the ferruginous sandstones and ironstones mainly contain low concentrations of U (<3mgkg⁻¹), the phosphatic pebbles contain up to 55mgkg⁻¹. It is, however, probable that the mass of the NSF, which in many cases contains disseminated radium, may contribute more to the overall level of radon emissions than the thin U-enriched phosphate horizons.

High levels of radon occur in both soil gas and houses underlain by Carboniferous Limestone in the UK as well as in caves and mines. There are 10% to more than 30% of houses built on the limestones that have radon concentrations greater than the UK action level (Appleton et al., 2000a). Much of the radon is thought to emanate from uranium- and radium-enriched residual soils that overlie the highly permeable limestones.

Chalk is a particularly abundant limestone in the south of England, but its radon emanation characteristics are different from the Carboniferous and Jurassic limestones. Chalk still retains its primary porosity, although most of the water and gas flow is through fissures. The proportion of dwellings with radon above the action level is much lower than over the Carboniferous Limestone, but higher levels of radon occur where the chalk is covered with conglomerate and residual clay-with-flint deposits.

Thick, permeable Cretaceous sand formations in southwestern England, including the glauconitic Lower and Upper Greensand and the Upper Lias Midford Sands, all emanate high levels of soil gas radon (mean values 20–48BqL⁻¹), and are characterized by a high proportion of houses above the action level (13 and 22% for the Upper Greensand and Midford Sands, respectively). In contrast, impermeable mudstones and clays in England and Wales are generally characterized by low to moderate soil gas radon (about 20BqL⁻¹) and less than 1% of homes exceed the action level.

Similar associations between high radon and Lower Carboniferous limestones, Namurian uraniferous and phosphatic black shales, and some granites and highly permeable fluvioglacial deposits have also been recorded in Ireland (Cliff & Miles, 1997).

In the Czech Republic, the highest indoor and soil gas radon levels are associated with the Variscan granites, granodiorites, syenites, and phonolites of the Bohemian massif. Syenites contain 12–20mgkg⁻¹U and the phonolites have 10–35mgkg⁻¹U and soil gas radon levels up to more than 450BqL⁻¹. High radon is also associated with Paleozoic metamorphic and volcanic rocks and also with uranium mineralization in the Příbram area (Barnet et al., 2002; Mikšová & Barnet, 2002)(Barnet et al., 2008; Barnet et al., 2010)

In Germany the highest radon occurs over the granites and Paleozoic basement rocks. Median soil gas radon for some granites ranges from 100 to 200BqL⁻¹ (Kemski et al., 2001)(Kemski et al., 2009). In contrast, the highest radon potential in Belgium is associated with strongly folded and fractured Cambrian to Lower Devonian bedrocks in which uranium preferentially concentrated in ferric oxyhydroxides in fractures and joints is considered to be the main source of radon (Zhu et al., 2001). In France some of the highest radon levels occur over peraluminous leucogranites or metagranitoids in a stable Hercynian basement area located in South Brittany (western France). These rocks are derived from uraniferous granitoids with average uranium contents of over 8mgkg⁻¹ (Ielsch et al., 2001). Soil gas and indoor radon concentrations were found to be controlled by lithology, structure, and uranium mineralization in India (Singh et al., 2002). High radon is associated with alum shale in both Sweden (Tell et al., 1993) and Belgium (Poffijn et al., 2002). In Korea, the mean values of soil gas radon concentrations were highest in granite gneiss and banded gneiss and lowest in soils over shale, limestone, and phyllite schist (Je et al., 1999). High indoor radon concentrations are associated with uranium-rich granite (Wang et al., 2005) in Zhuhai City, China with the two highest indoor radon concentrations found in workshops producing potable water from granite groundwater containing high radon concentrations (1300-1980 Bq L⁻³). (Chen et al., 2005). Radon from thermal spring waters in granite terrains contributes a major proportion of indoor radon in hot spring hotels located in Guangdong province, China (Song et al., 2011). High concentrations of radon and radium are also found in groundwaters associated with granite in Fujian province, China ((Zhuo et al., 2001)). Wang et

al (2009, 2011)(Wang et al., 2009; Wang et al., 2011) found that there is a strong correlation between radioactivity level and geological lithology with the highest exhalation rates and soil gas radon concentrations being associated with biotite granite (151 kBq m^{-3}) and lower concentrations with weathered granite (38 kBq m^{-3}) and Quaternary sediments (7 kBq m^{-3}). The impact of unconsolidated deposits mainly reflects their permeability. For example, peat and lacustrine clays strongly reduce radon potential associated with the underlying bedrock, whereas permeable sand and gravel and river terrace deposits tend to enhance radon potential. In Sweden fragments and mineral grains of uranium-rich granites, pegmatites, and black alum shales are dispersed in till and glaciofluvial deposits leading to high radon in soils and dwellings, especially when the glaciofluvial deposits are highly permeable sands and gravels (Clavensjö & Åkerblom, 1994). In Norway, high radon levels are associated with highly permeable fluvioglacial and fluvial deposits derived from all rock types and also with moderately permeable unconsolidated sediments (mainly basal till) containing radium-rich rock fragments (Smethurst et al., 2008; Sundal et al., 2004).

XIV. Administrative and Technical Responses

A. Environmental Health

Environmental health responses include provision of guidance for radon limitation including recommendations for dose limits and action levels, establishment of environmental health standards for houses and workplaces, and enforcement of Ionizing Radiations Regulations to control exposure to radon in workplaces (Åkerblom, 1999; Appleton et al., 2000b; NRPA, 2000;(WHO, 2009)). National authorities are recommended to set a reference level for radon that represents the maximum accepted radon concentration in a residential dwelling. Remedial actions may be recommended or required for homes in which radon exceeds the reference level. Various factors such as the distribution of radon concentrations, the number of existing homes with high radon concentrations, the arithmetic mean indoor radon level and the prevalence of smoking are usually considered when setting a national radon reference level. WHO (2009) proposed a reference level of 100 Bq m^{-3} to minimize health hazards due to indoor radon exposure although if this concentration cannot be reached, the reference level should not exceed 300 Bq m^{-3} . This represents approximately 10 mSv per year according to recent calculations and recommendations by the International Commission on Radiation Protection (ICRP, 2007, 2009) which also recommends 1000 Bq m^{-3} as the entry point for applying occupational radiological protection requirements. There are substantial variations in action levels (or their equivalents) in countries that perceive a radon problem. International and national recommendations for radon limitation in existing and future homes, given as the annual average of the gas concentration in Bq m^{-3} , range from 150 to 1000 for existing dwelling and from 150 to 250 for new dwellings (Åkerblom, 1999). The majority of countries have adopted 400 and 200 Bq m^{-3} , respectively, for the two reference levels(HPA, 2009).

The reasons for these different reference levels appear largely historical but are also due to a combination of environmental differences, different construction techniques, and varying levels of political and environmental concern. There would be advantages in harmonizing standards because the existence of different levels may lead to confusion among the public. The ICRP considers that one common international standard is unlikely to be achieved, and that this is less important than achieving reasonable reductions in radon levels in radon prone areas.

In addition to variations in house radon standards between countries, there also appears to be some variation in standards applied within the field of radiological protection. For example, some observers have suggested that if the highest natural radon levels recorded in some houses in Cornwall and Devon were reached, for example, at a nuclear installation, it would be closed down immediately. Equally, it is reported that all houses in the Chernobyl area had to be evacuated under Soviet law if they were contaminated to a level equivalent to the current UK action level of 200 Bq m^{-3} . However, international and national radiological protection authorities are united in acknowledging the need for a distinction in the ways radiation is approached in these different circumstances.

Recommendations differ from country to country. In the UK, testing for radon is recommended by government in radon Affected Areas where more than 1% of dwellings exceed the action level of 200Bq m^{-3} . In 2010, HPA guidance on radon gas concentrations introduced a new 'target level' of 100 Bq m^{-3} above which remediation work should be seriously considered especially by high risk groups such as smokers and ex smokers.. In contrast, the U. S. EPA recommends that all homes be tested for radon because (1) high levels of indoor radon have been found in every state, (2) radon levels vary so much from place to place, and (3) because dwellings differ so radically in their vulnerability to radon. The U. S. EPA estimates that 1 in every 15 homes has radon levels higher than 4pCi L^{-1} (148Bq m^{-3}), the level above which the EPA recommends that corrective action is taken. The U.S. EPA recommends that no action is required below 150Bq m^{-3} , action within a few years between 170 and 750Bq m^{-3} , urgent action between 750 and 7500Bq m^{-3} , and immediate action above 7500Bq m^{-3} .

The European Commission Recommendation (2001/928/Euratom) on the protection of the public against exposure to radon in drinking water supplies recommends 1000Bq/L as an action level for public and commercial water supplies above which remedial action is always justified on radiological protection grounds. Water supplies that support more than 50 people or distribute more than 10m^3 per day, as well as all water that is used for food processing or commercial purposes, except mineral water, are covered by the Europe Commission Recommendation. The 1000Bq/L action level also applies to drinking water distributed in hospitals, residential homes, and schools and should be used for consideration of remedial action in private water supplies. The U. S. EPA recommends that states develop multimedia mitigation (MMM) programs to address the health risks from radon in indoor air while individual water systems should reduce radon levels in drinking water to 148Bq L^{-1} (4000pCi L^{-1}) or lower. The EPA is encouraging states to adopt this option because it is the most cost-effective way to achieve the greatest radon risk reduction. If a state chooses not to develop an MMM program, individual water systems would be required to either reduce radon in their system's drinking water to 11Bq L^{-1} (300pCi L^{-1}) or develop individual local MMM programs and reduce levels in drinking water to 148Bq L^{-1} (4000pCi L^{-1}). The regulations will not apply to private wells, because the EPA does not regulate them. A guideline value of $2\mu\text{g L}^{-1}$ uranium (equivalent to approximately 0.02Bq L^{-1}) is recommended by the World Health Organization (WHO), although this is based on its toxicity, which is more detrimental to health than its radioactivity (WHO, 1996). The U. S. EPA established maximum contaminant levels (MCL) of $30\mu\text{g L}^{-1}$ uranium, and 0.185Bq L^{-1} for radium-226 and radium-228 in community water supplies. No specific value for uranium is given in the EU Directive for drinking water (CEC, 1998), which establishes a 0.1mSv y^{-1} total indicative dose guidance level for radionuclides, excluding tritium, ^{40}K radon, and radon decay products. Action levels ranging from 7.4 to 160Bq L^{-1} uranium in drinking water have been reported for Austria, Finland, and France.

B. Radon Monitoring

The overall aim of most countries that have identified a radon problem is to map radon-prone areas and then identify houses and workplaces with radon concentrations that exceed the radon reference level. In the UK, for example, radon affected areas are delineated by measuring radon in a representative sample of existing dwellings. Householders are then encouraged to have radon measured in existing and new dwellings in affected areas and local authority environmental health departments are generally responsible for ensuring that radon in workplaces is monitored in appropriate areas.

C. Protective Measures

Provisions have been made in the building regulations to ensure that new dwellings are protected against radon where a significant risk of high radon concentrations in homes has been identified on the basis of house radon surveys. Nine European countries (Czech Republic, Denmark, Finland, Ireland, Latvia, Norway, Slovak Republic, Sweden, and the UK) have regulations and guidelines for construction requirements to prevent elevated radon concentrations in new buildings. Austria, Germany, Greece, and Switzerland plan to introduce such regulations. In most of the countries with regulations, enforced radon protection in new buildings is specified in the national building codes. Implementation of regulations is normally shared by national and local authorities. Eight Europe-

an countries (Czech Republic, Denmark, Finland, Ireland, Norway, Slovak Republic, Sweden, and the UK) have regulations and guidelines for radon prevention in the planning stages of new development (e.g., where construction permits are applied for dwellings, offices, and factories). Austria and Germany are considering the introduction of guidance and/or regulations for dealing with radon at the planning stage. In the Czech Republic, Ireland, Slovak Republic, and Sweden regulations require an investigation of radon risk at construction sites before building is permitted (Åkerblom, 1999). Cost-effectiveness analysis in the UK suggest that radon protective (preventive) measures, such as a sealed membrane under the ground floor in new homes would be justified in all areas (HPA, 2009).

D. Remedial Measures

In the UK, owners of workplaces may be forced to carry out remedial measures whereas householders in dwellings with radon above the action level are generally only advised to take action to reduce the radon level. Guidance on reducing radon in dwellings is provided, but the cost of installing remedial measures in a dwelling is normally the householder's responsibility. Grant aid may be available.

The principal ways of reducing the amount of radon entering a dwelling are similar to those used for protective measures in new dwellings. These are

1. Install an airtight barrier across the whole of the ground floor to prevent radon getting through it and also seal voids around service inlets
2. Subfloor ventilation of underfloor cavities, i.e., drawing the air away from underneath the floor so that any air containing radon gas is dispersed outside the house
3. Subfloor depressurization (radon sump)
4. Positive pressurization (i.e., pressurize the building in order to prevent the ingress of radon)
5. Ventilation (i.e., avoid drawing air through the floor by changing the way the dwelling is ventilated)

In the United States, the cost of radon mitigation in a typical home ranges from about \$500 to about \$2500. Fitting radon resistant measures at the time of construction would cost \$350–\$500. Similar costs apply in the UK.

Radon gas may be easily removed from high-radon groundwaters by aeration and filter beds will remove daughter products. Various aeration technologies are available including static tank, cascade, or forced aeration in a packed tower. Radon removal technologies used in the United States include removal of ^{222}Rn by spray jet aeration, packed tower aeration, and multistage bubble aeration. Packed tower aeration is simple and cheap and is recommended for large drinking water supplies. Removal of ^{222}Rn by granular activated carbon is efficient but ^{238}U decay products, including U, Po, Bi, and Pb (^{210}Pb), are adsorbed onto the activated carbon, which produces a disposal problem. The U. S. EPA recommends that the most practical treatment methods for radionuclide removal are ion exchange and lime-soda softening for radium, aeration and granular activated carbon for radon, and anion exchange and reverse osmosis for uranium.

Monitoring and remediation in existing homes are not cost-effective at present in the UK but might become so in areas with mean indoor radon concentrations of 60 Bq m^{-3} or above if the UK Action Level was reduced from 200 to 100 Bq m^{-3} (HPA, 2009).

See Also the Following Chapters

Chapter 2 (Natural Distribution and Abundance of Elements) □ Chapter 9 (Volcanic Emissions and Health) □ Chapter 21 (Environmental Epidemiology)

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Table I The uranium-238 decay series

Nuclide	Principle mode of decay	Half-life
²³⁸ U	α	4.5×10^9 years
²³⁴ Th	β	24.1 days
²³⁴ Pa	β	1.2 minutes
²³⁴ U	α	2.5×10^5 years
²³⁰ Th	α	7.5×10^4 years
²²⁶ Ra	α	1,602 years
²²² Rn	α	3.8 days
²¹⁸ Po	α	3.1 minutes
²¹⁴ Pb	β	26.8 minutes
²¹⁸ At	α	1.5 seconds
²¹⁴ Bi	α	19.9 minutes
²¹⁴ Po	α	1.6×10^{-4} seconds
²¹⁰ Tl	β	1.3 minutes
²¹⁰ Pb	β	22.6 years
²¹⁰ Bi	β	5.0 days
²¹⁰ Po	α	138.4 days
²⁰⁶ Tl	β	4.2 minutes
²⁰⁶ Pb	Stable	Stable

Table II The thorium-232 decay series

Nuclide	Principle mode of decay	Half-life
²³² Th	α	1.4×10^{10} years
²²⁸ Ra	β	5.8 years
²²⁸ Ac	β	6.1 hours
²²⁸ Tl	α	1.9 years
²²⁴ Ra	α	3.7 days
²²⁰ Rn	α	55.6 seconds
²²⁶ Po	α	0.15 seconds
²¹² Pb	β	10.6 hours
²¹² Bi	α 36%	60.5 minutes
	β 64%	
²¹² Po	α	3.0×10^{-7} seconds
²⁰⁸ Tl	β	3.1 minutes
²⁰⁷ Pb	Stable	Stable

Table III. Units of measurement of radioactivity and dose

Quantity	Unit	Purpose	Comments
Activity	becquerel (Bq)	Measure activity of a radioactive material (solid or gas); The International System of Units (SI) definition of Activity.	1 Bq = 1 Atomic disintegration per second
	curie	In the USA, the activity (rate of decay) of ^{222}Rn is expressed in units called curies.	The curie is based on the rate of decay of one gram of ^{226}Ra or 3.7×10^{10} disintegrations per second.
	pico curies (pCi)		1 pCi = one trillionth of a Curie; 0.037 disintegrations per second, or 2.22 disintegrations per minute.
Radioactivity in air or water	becquerels/m ³ (Bq m ⁻³)	Measure average concentration of radon gas in building or in soil air. Bq/L used to measure radon in water.	Average level of radon in houses in Great Britain is 20 Bq m ⁻³ ; in Sweden 108 Bq m ⁻³
	picuries/L ⁻¹ (pCi L ⁻¹)	Unit used in USA	Average level of radon in houses in USA is 1.24 pCi L ⁻¹ equivalent to 46 Bq m ⁻³
Absorbed dose	gray (Gy)	Measure energy per unit mass absorbed by tissue	1 Joule of energy absorbed by 1 kg of tissue
	rad	Old unit of absorbed dose	1 rad = 0.01 Gy
Equivalent dose	sievert (Sv)	Measure of absorbed doses caused by different types of radiation	Absorbed dose weighted for harmfulness of different radiations.
	roentgen man equivalent (rem)	Old measure of absorbed dose	The rem is being replaced by the Sievert, which is equal to 100 rem.

Table IV Principal decay properties of radon (^{222}Rn) and short-lived decay products (after Green et al., 1992)

Radionuclide		Half-life	Main radiation energies and intensities					
			α		β		γ	
			MeV	%	MeV	%	MeV	%
^{222}Rn	Gas	3.824 day	5.49	100	-	-	-	-
^{218}Po	Solid	3.11 min	6.00	100	-	-	-	-
^{214}Pb	Solid	26.8 min	-	-	1.02	6	0.35	37
					0.70	42	0.30	19
					0.65	48	0.24	8
^{214}Bi	Solid	19.7 min	-	-	3.27	18	0.61	46
					1.54	18	1.77	16
					1.51	18	1.12	15
^{214}Po	Solid	1.64×10^{-4} s	7.69	100	-	-	-	-

Table V. Estimated annual absorbed doses to adult tissues from ^{222}Rn and its short-lived progeny for domestic ^{222}Rn concentration of 20 Bq m^{-3} (NAS, 1998).

Tissue	Annual dose ($\mu\text{Gy y}^{-1}$)
Lung	500
Skin*	50-1,000
Red bone marrow	0.5-6
Bone surface	0.4-4.4
Breast	1.2-1.5
Blood	1.1
Liver	2.5
Kidney	14.4

* basal cells at $50\mu\text{m}$ in exposed skin

TABLE VI. Fatal Lifetime Lung Cancer Risks for Lifetime Radon Exposure at 200Bqm^3 Based on BEIR VI Models

	<i>Risk (%)</i>
General population	3–5
Smokers	10–15
Non-smokers	1–3

From NRPB, 2000.

TABLE VII. Cumulative absolute risk of death from lung cancer to age 75 years in the UK by long-term average residential radon concentrations and smoking history (adapted from Table 4.3(HPA, 2009))

Long-term average radon concentration (Bq m ⁻³)	Cumulative risk of death from lung cancer to age 75 years (%)		
	Lifelong non-smoker	Ex-cigarette smoker stopped at age 50	Continuing cigarette smoker
0	0.41	5.5	14.7
21*	0.42	5.7	15.2
100	0.47	6.4	16.9
200	0.53	7.2	19
400	0.66	8.9	23
800	0.92	12.2	30.5

* mean UK long-term average residential radon concentration

Table VIII Sources of radiation in Great Britain for average person in the UK (NRPB, 1989)

Source	% Annual dose
	87.0
<i>Natural sources</i>	
Radon (^{222}Rn) gas from the ground	47.0
Thoron (^{220}Rn) from the ground	4.0
Gamma rays from the ground and building materials	14.0
Food and drink	12.0
Cosmic rays	10.0
<i>Artificial sources</i>	
Medical	12.0
Nuclear discharges	0.1
Work	0.2
Fallout	0.4
Miscellaneous	0.4

Table IX. Criteria used in Sweden for classifying high-radon and low-radon ground (after Clavensjö and Åkerblom, 1994)

Bed-rock or overburden	^{226}Ra (Bq kg $^{-1}$)	^{222}Rn in soil gas (Bq L $^{-1}$)
<i>High radon ground.</i>		
Bare rock	>200	not relevant
Gravel, sand, coarse till	>50	>50
Sand, coarse silt	>50	>50
Silt	>70	>60
Clay, fine till	>110	>120
<i>Low radon ground</i>		
Bare rock	<60	
Gravel, sand, till	<25	<20
Silt	<50	<20
Clay, fine till	<80	<60

Table X. Czech Republic radon risk classes based on radon in soil gas and rock-overburden permeability (after Barnett, 1994)

Radon Risk	Rock-Overburden Permeability		
	High	Medium	Low
	Radon concentration in soil gas (Bq L^{-1})		
High	>30	>70	>100
Medium	10-30	20-70	30-100
Low	<10	<20	<30

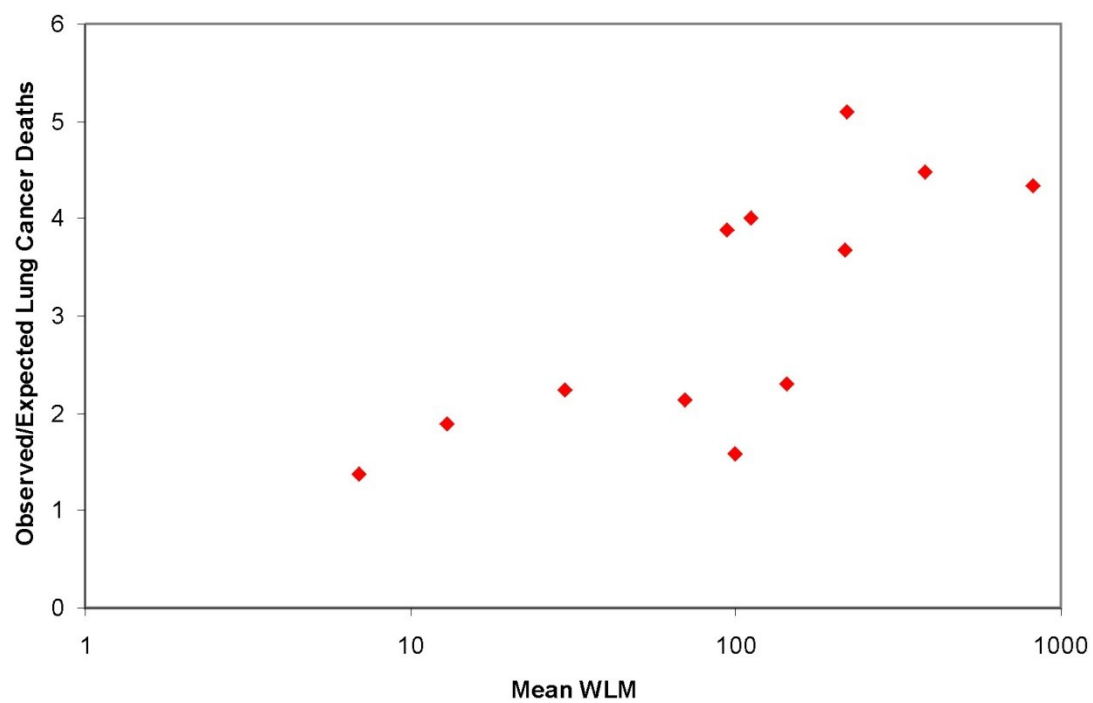


FIGURE 1 Excess mortality from lung cancer among miners exposed to high levels of radon. WLM = Working Level Month, a unit of exposure to radon (Data from NRPB, 2000).

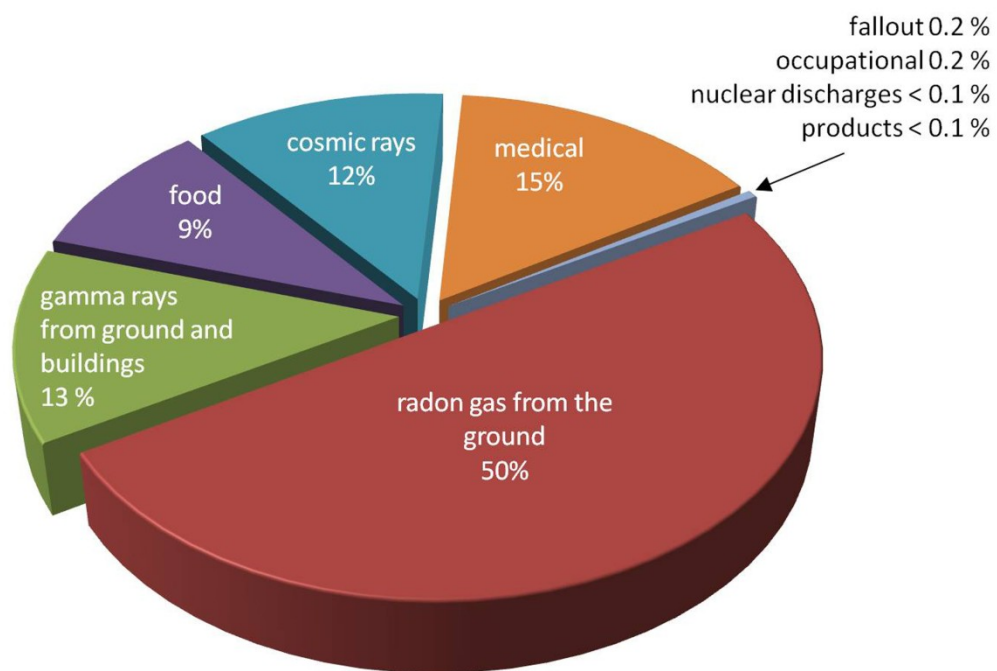


FIGURE 2 Sources of radiation exposure contributing to average effective dose in the UK. Other = occupation 0.3%, fallout 0.2%, nuclear discharges <0.1%, and products <0.1%. (Data from Watson et al. 2005)

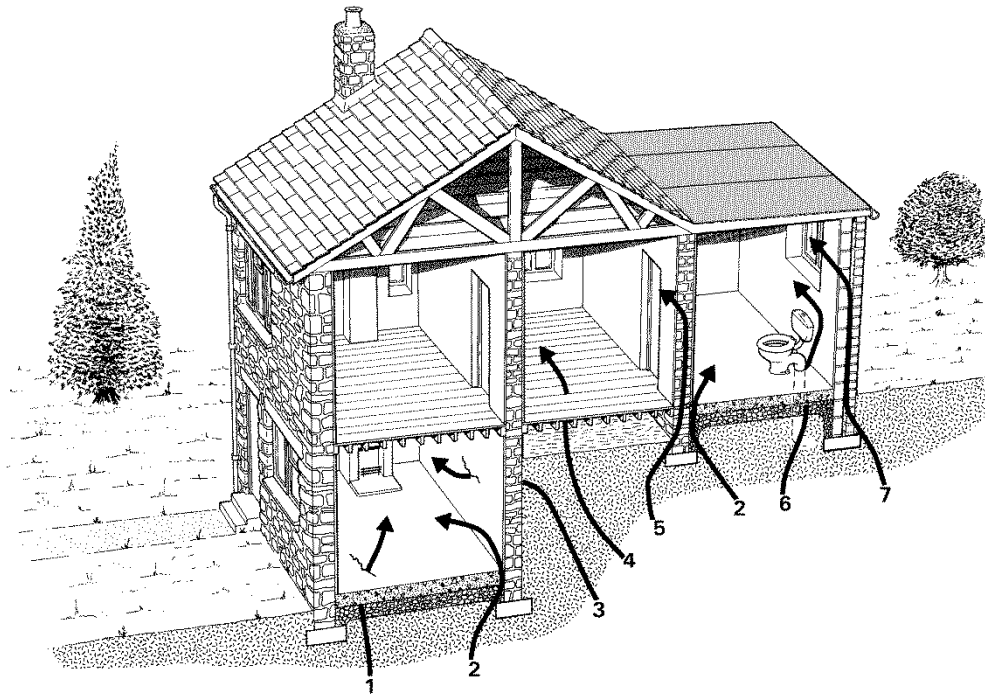


FIGURE 3 Routes by which radon enters a dwelling. (Reproduced from BR211, BRE 1999 with permission from C. Scivyer, BRE)

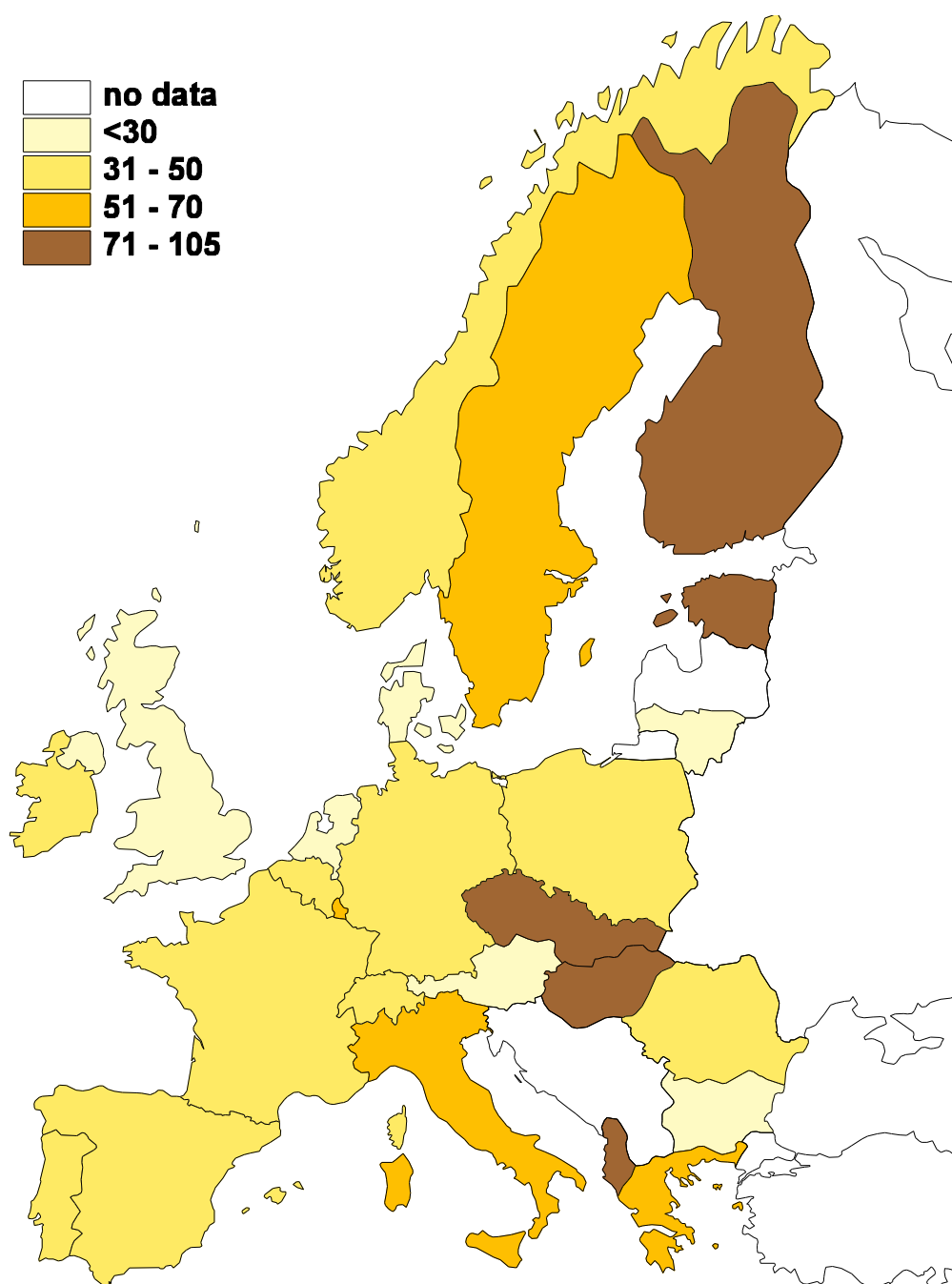


FIGURE 4 Geometric mean radon concentrations (Bqm⁻³) indoors in Europe. (Compiled from data in UNSCEAR, 2000.)

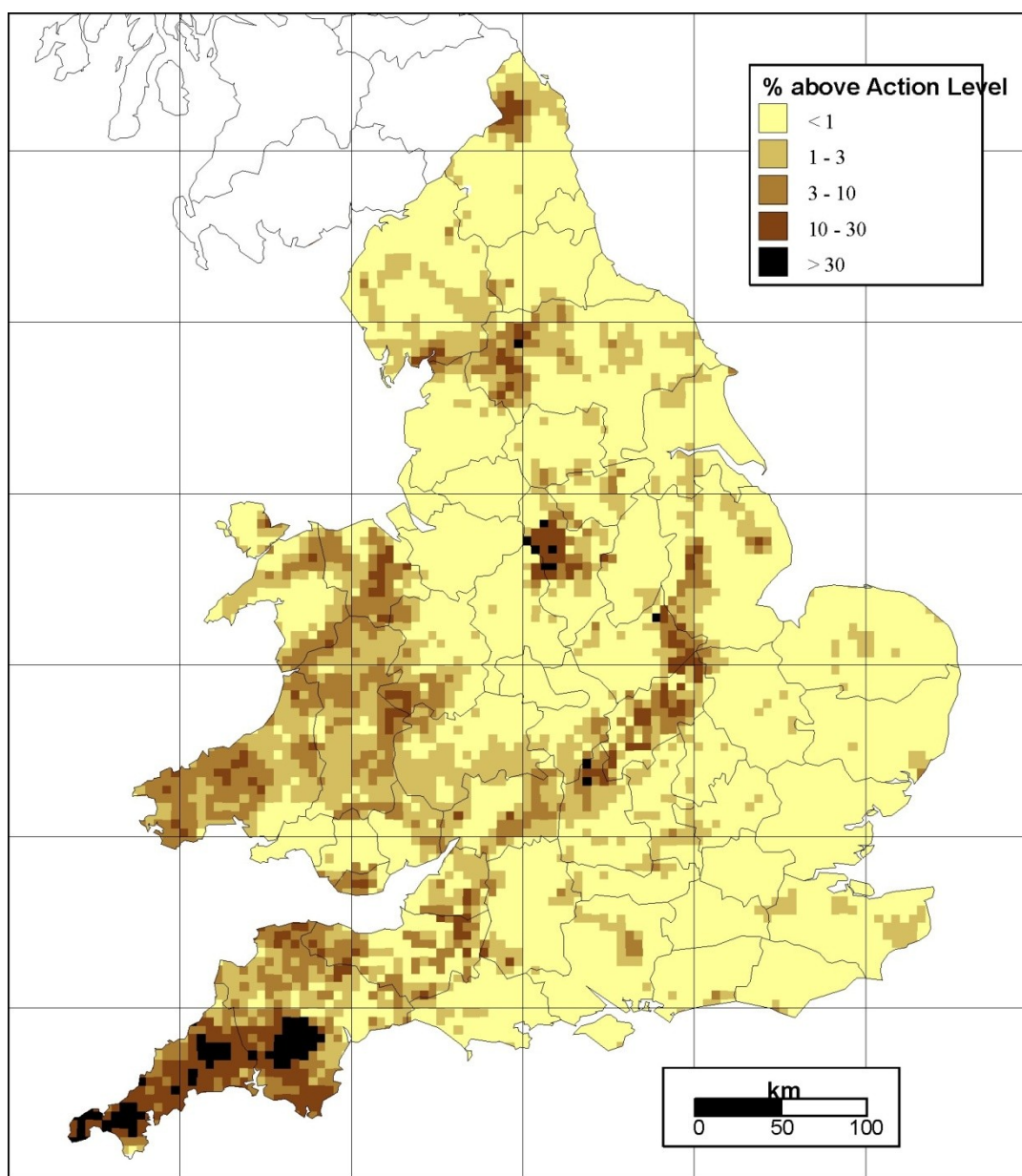


FIGURE 5 Estimated proportion of homes exceeding the action level in each 5-km grid square of England and Wales. (Adapted from Figure 2-2 in Appleton et al., 2000a.)

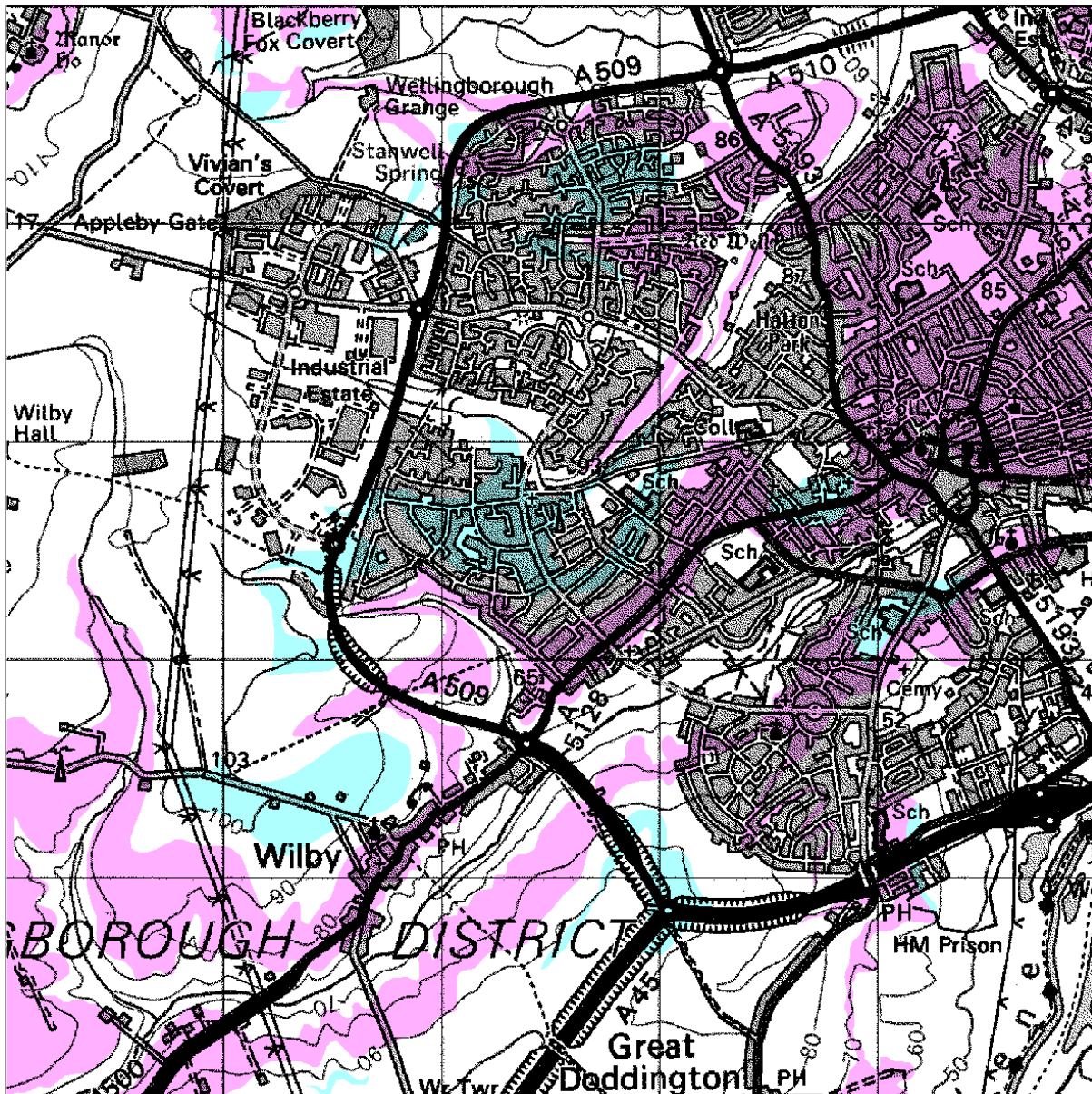


FIGURE 6 Geological radon potential map of the 5-km grid square (485265) that encompasses the western sector of Wellingborough, England. The 1:50,000 scale map illustrates the distribution of geological units with <3% (white), 3–5% (blue), and 10–20% (pink) of dwellings above the UK radon action level. The 5-km grid square has an average radon potential of 3.9% (NRPB 1998 data). (Topography based on Ordnance Survey 1:50,000 Scale Colour Raster data with permission of The Controller of Her Majesty's Stationery Office Crown Copyright. Ordnance Survey Licence number GD272191/2004.)

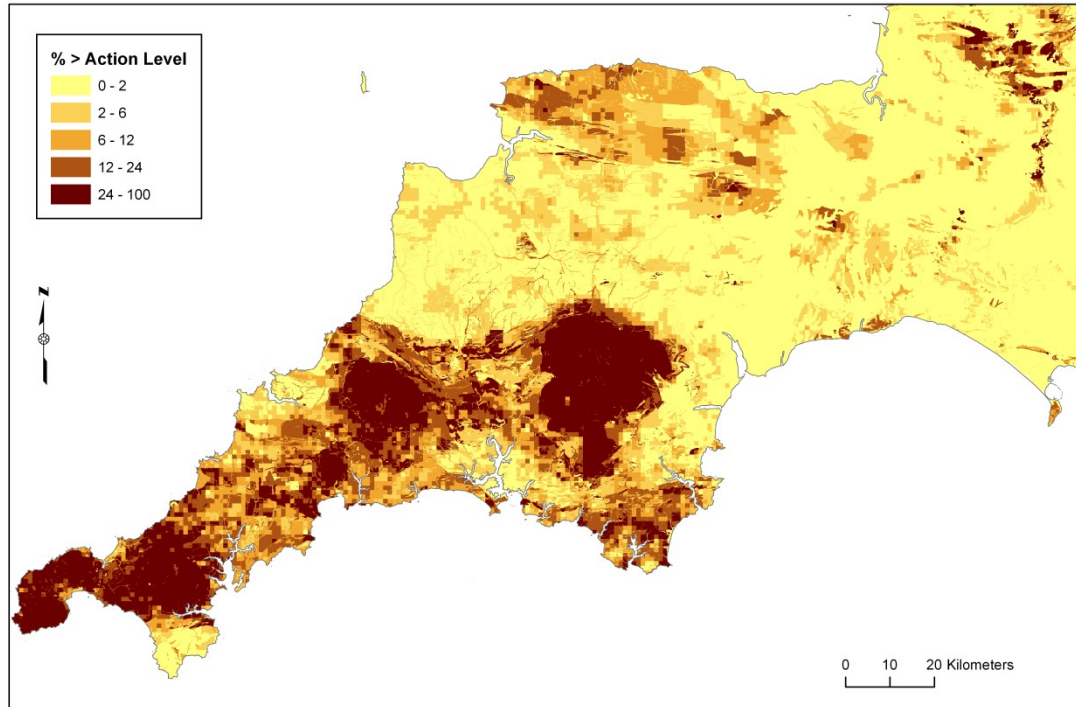


FIGURE 7. Radon potential map of southwest England based on geology and indoor radon measurements (adapted from Figure 8 in Miles and Appleton, 2005)

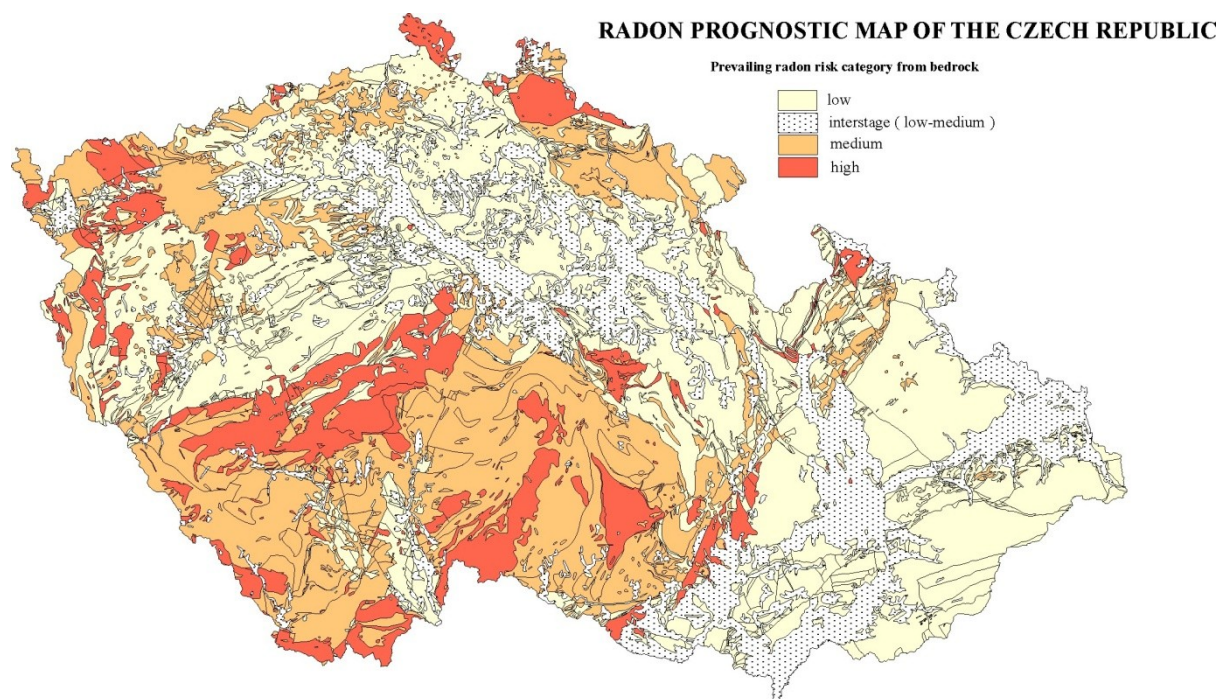


FIGURE 8 Radon prognostic (risk) map of the Czech Republic. (Reproduced with permission from C. Barnet, Czech Geological Survey.)

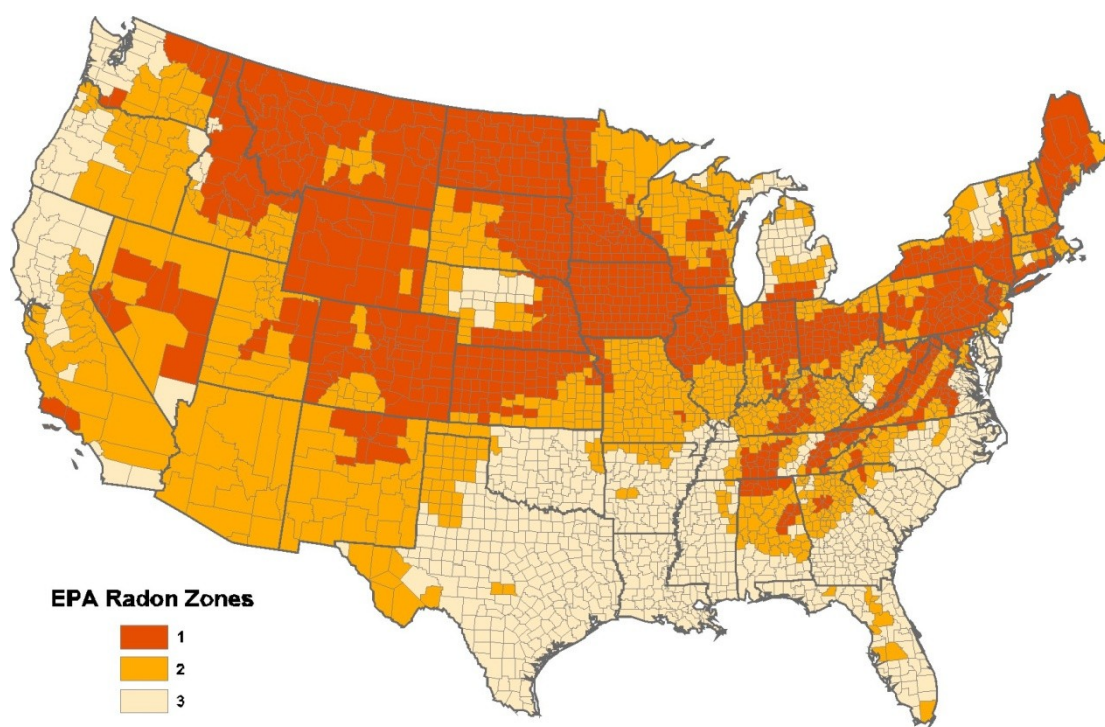


FIGURE 9 U. S. EPA Map of Radon Zones (excluding Alaska and Hawaii). Zone 1, 2, and 3 counties have a predicted average indoor radon screening concentration of >148 , $74\text{--}148$, and $<74\text{Bqm}^{-3}$, respectively. (Map based on state radon potential maps available at <http://www.epa.gov/radon/zonemap.html>; state and county boundaries SRI ArcUSA 1:2M.)

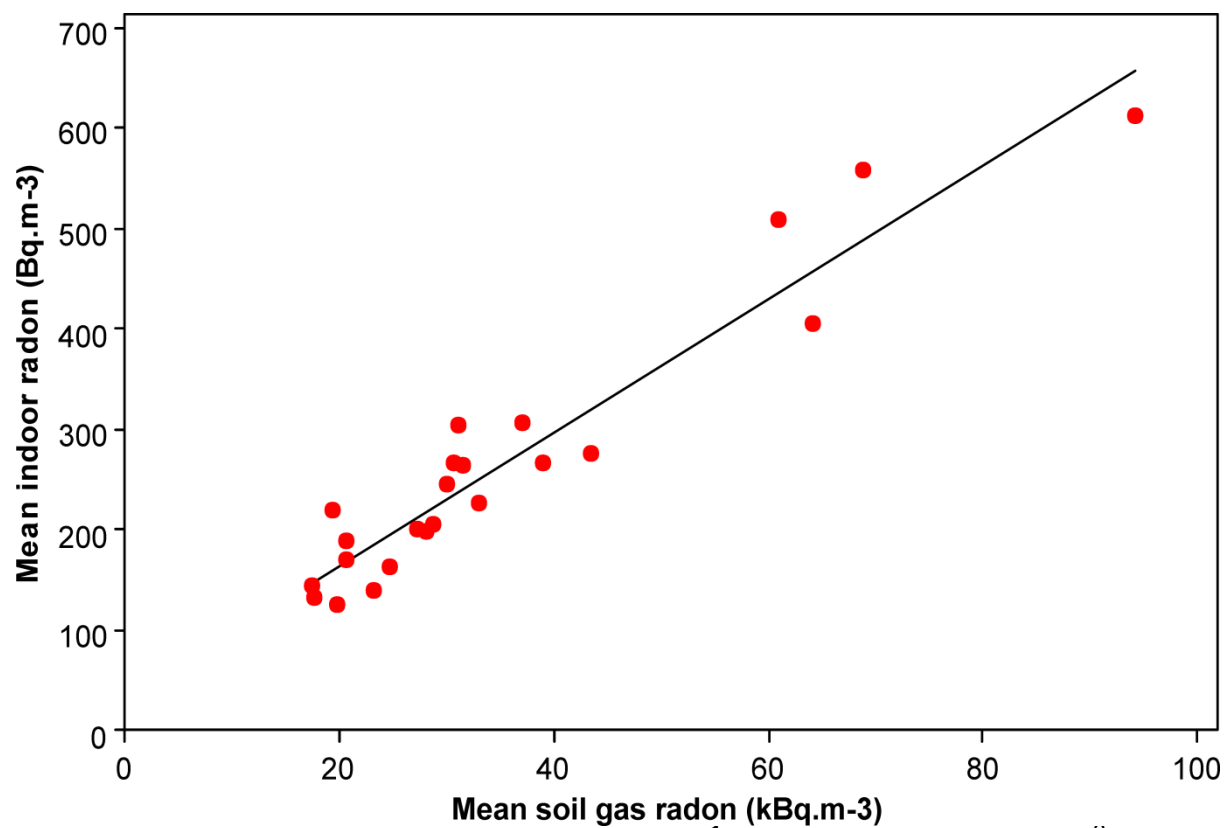


FIGURE 10 Relationship between average soil gas radon (BqL^{-1}) and average indoor radon (Bqm^{-3}) for major rock types of the Czech Republic. (Based on data in Table 1, Barnet et al., 2010.)

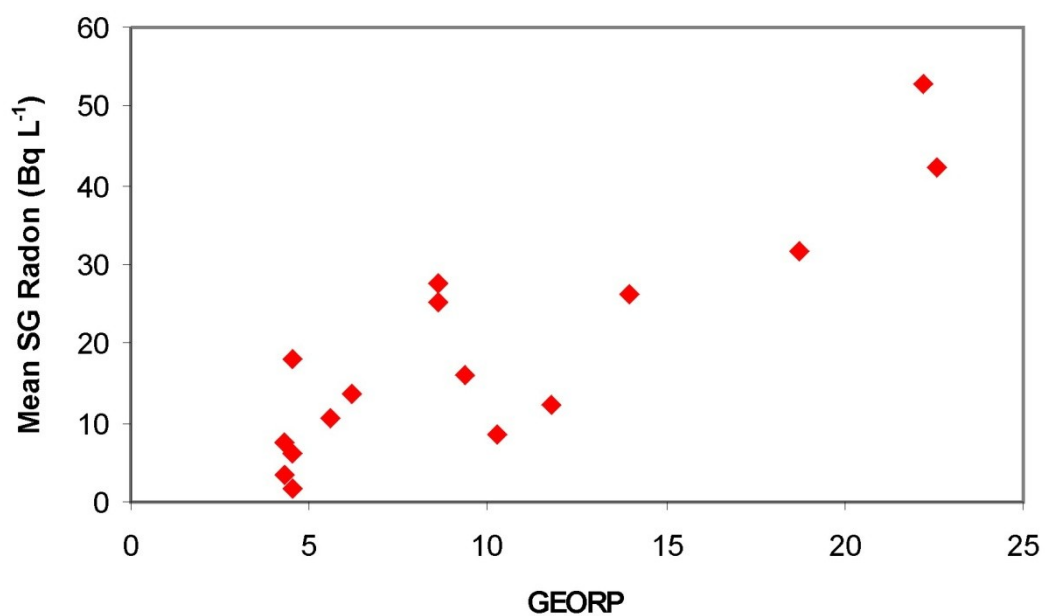


FIGURE 11 Relationship between average soil gas radon concentration (Bq L^{-1}) and the geological radon potential (GEORP = estimated proportion of dwellings exceeding the UK radon action level, 200Bq m^{-3}). Data for dwellings sited on the Jurassic Northampton Sand Formation grouped by 5-km grid square). (Reproduced from Appleton et al, 2000a.)

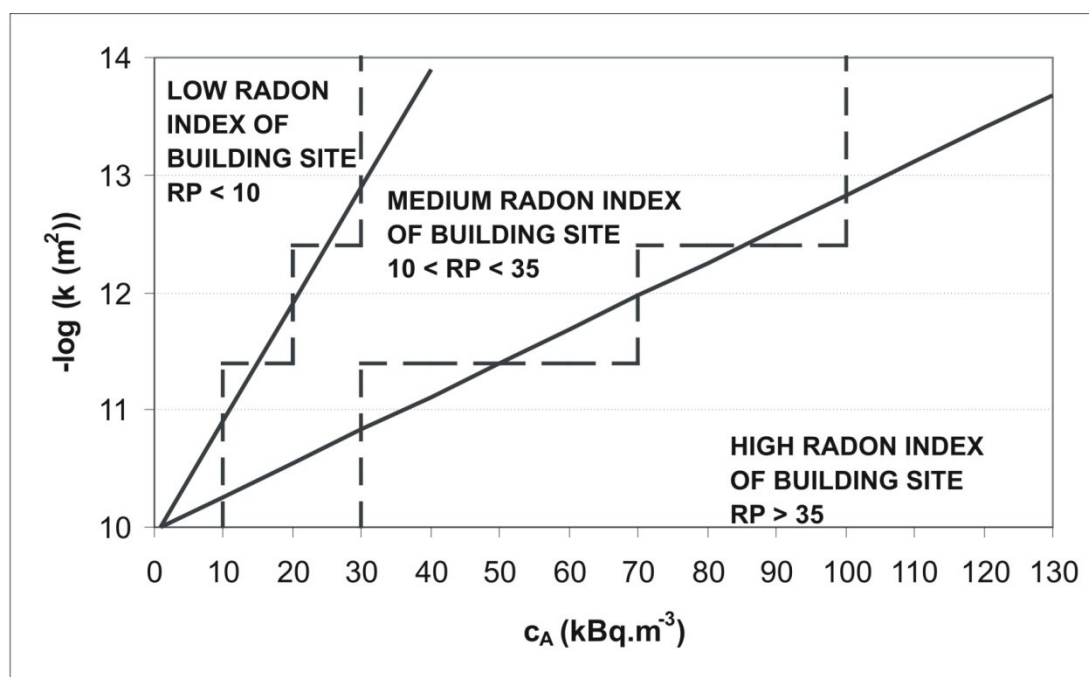


FIGURE 12. Czech radon potential classification for assessing the radon risk of building sites (copy of Figure 6-2, Barnet et al. 2008; reproduced with permission from I. Barnet, Czech Geological Survey)