2. Occurrence, Exposure and Analysis

2.1 Occurrence

(*a*) *Sources*

Radium-226 in the earth's crust is the main source of radon-222 in the global environment. Radon is ubiquitous throughout the geosphere, biosphere and atmosphere, since radium-226 is present everywhere and the gaseous radionuclide is highly mobile. Radium-226 concentrations in soils range over several orders of magnitude but are generally between 10 and 50 Bq/kg, with an estimated average concentration of 25 Bq/kg (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982; McLaughlin, 1986). In sea-water, the concentration of radium-226 is about four to five orders of magnitude lower than that in soils (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Ocean sediments contain large stocks of radium-226; thus, deeper sea-water has higher concentrations of radium-226 and radon than surface sea-water. However, this source does not contribute substantially to environmental concentrations of radon (Harley, 1976; Miyake *et al.*, 1980).

Typical rates of exhalation of radon from various soils throughout the world range from about 0.0002 to 0.07 Bq/m^2 per sec; worldwide release of radon from soils is 5–10 × 10^{19} Bq/year. The oceans contribute less — about 9 × 10^{17} Bq/year (Harley, 1976; United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

Numerous other, generally minor sources contribute radon to the world inventory, although some provide quite significant amounts to local, regional or national environmental levels. Plants can increase the amount of radon released into an area. Radon from the burning of natural gas and coal contributes only minor amounts, estimated to be about

10^{14} Bq/year and 10^{13} Bq/year, respectively. Similarly, the total amount of radon released from uranium mines, mills and tailings is relatively small — only about 10^{15} Bq/year. Geothermal energy sources, phosphate mining and milling, fertilizer utilization, mineral extraction industries and construction materials are also minor sources of radon in the environment (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Surface-waters generally contain low concentrations of radon, but levels as high as 37×10^6 Bq/m^3 have been reported in some ground-waters (Hess *et al.*, 1985a).

(*b*) *Occurrence and transport in soil*

Radon formed in rocks and soils is released to the surrounding water or air only partially. A fraction of the radon makes its way into pore spaces and is transported into nearby surroundings. Diffusion, convection and general flow of air or water are the principal mechanisms for transport of radon. High soil porosity increases the diffusion rate. Also, modest amounts of moisture enhance the release of radon, whereas high moisture levels decrease it because of slowed diffusion (Tanner, 1980; United Nations Scientific Committee on the Effects of Atomic Radiation, 1982; Tanner, 1986).

Once radon has entered the water or air phase within soil, transport mechanisms include diffusion, percolation and mechanical and convective flow. Little radon migrates by diffusion over long distances; for example, it has been estimated that radon decays by about 90% after diffusion through about 5 m of air, 5 cm of water or about 2 m of soil. Transport beyond these distances involves other mechanisms (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

Mechanical forces, such as earth tides and earthquakes, which cause changes in pore spacing, may also contribute to the transportation of radon in bedrock and soil. Thermal and pressure flows are also probably significant factors in the transport of radon within soils. The movement of ground waters through percolation and lateral flows underground is probably a major factor in the movement of radon deep within the ground (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

The radon content of ground-waters is normally related to the radium-226 concentration in the surrounding rock. Consequently, high concentrations are found in groundwaters in the vicinity of uranium ore bodies, granite, pegmatite, syenite and porphyry. These geological formations often contain relatively high concentrations of uranium-238 and radium-226 (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). In an investigation of the levels of radon in ground-water in Maine, USA, high concentrations of radon were found in granites and adjacent sedimentary rocks that have undergone great stress, change and fracturing. Low concentrations were found in groundwater in nongranitic areas where the rocks had not undergone much metamorphic change (Hess *et al.*, 1980).

Soils contain various amounts of air, depending on their permeability and density. Radon enters the soil gas by diffusion from nearby soil particles or as a result of migration from more distant radon-rich materials. The concentration of radon in soil gas decreases nearer the surface as the soil gas escapes to the open air above ground. Corresponding theoretical models have been developed. The concentration of radon in soil gas is affected by meteorological factors, such as barometric pressure, temperature, humidity, wind speed and precipitation. In general, factors that influence the concentration of soil gas also influence the exhalation rate of radon from the ground, but they usually affect it in the opposite direction. For example, rain, snow and increased atmospheric pressure reduce the exhalation rate. Consequently, the radon concentration in soil is maximal in winter, when the ground is frozen, and during rainy periods. High wind speeds and temperature increase exhalation rates, which decrease soil gas concentrations. Soil gas and exhalation rates can vary substantially among soils. Also, diurnal variations of exhalation rates have been observed by some investigators (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

(*c*) *Occurrence and dispersion in air*

(i) *Outdoor air*

The transport and dispersion of radon in air depends on the vertical temperature gradient, the direction and strength of the wind, and air turbulence. Concentrations of radon decay products are also affected by precipitation. Most of the air mass, water vapour and dust are found in the troposphere (75%), and, under normal conditions of turbulence, most (>99%) radon and its decay products are also found in the troposphere. Radon concentrations in the air vary daily and seasonally. Maximal concentrations occur in late summer, and minimal levels are observed in winter or spring. Generally, radon levels reach their maximum in the early morning and their minimum at noon or in the afternoon (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

The release of radon into the atmosphere from sources such as piles of uranium mill tailings, uranium mine ventilation systems, phosphogypsum piles and other point and area sources can be modelled by dispersion techniques. However, beyond 1–2 km from the source, the additional contribution of piles or vents may not be distinguishable within the ambient radon level (US Environmental Protection Agency, 1983, 1985, 1986a).

(ii) *Indoor air*

Sources that contribute radon inside structures, buildings and other confined spaces are of particular significance with regard to the exposure and health of humans. In the outside air, wind and temperature gradients act quickly to reduce the concentrations of radon emanating from the ground. Inside confined areas, however, low rates of air change can result in a build-up of radon and its decay products to levels tens of thousands of times higher than those typically observed outdoors. Radon in indoor spaces may originate from exhalation from rock and soils around the building or from construction materials used in walls, floors and ceilings. Radon may also be released from materials brought into the building, such as natural gas and well-water that contain high concentrations of radon. Radium-226 is the primary source of radon in these situations (Bruno, 1983; Nero, 1983; US Environmental Protection Agency, 1986b).

The relative contributions of the sources listed above to radon concentrations inside buildings vary. In the USA, building materials, ground-water and natural gas usually contribute much less than soil to the total radon level inside a building, except when ground-water contains large concentrations of radon (>400 000 Bq/m^3) or building materials contain high levels of radium-226 (>100 Bq/kg) (Bruno, 1983; Hess *et al.*, 1987; Sextro *et al.*, 1987). [The Working Group noted that, in Europe, where building practices are substantially different from those in the USA, building materials may be a generally more important source, particularly in areas where there are very low concentrations of radon in the soil gas.]

Radon originating in rocks and soil under a structure may enter through a variety of apertures. Some of the common entry points within a building include foundation joints, cracks in floors and walls, openings in sills above hollow block walls, sump holes, drains and piping and electrical penetrations in walls and floors (US Environmental Protection Agency, 1986b).

In the 1960s, sand tailings from the processing of uranium ores were widely used in building construction throughout the western USA. The tailings were used in concrete and as back fill around foundations, as bedding under foundations and in numerous other applications. As a consequence, some houses contained radon concentrations of up to about 4000 Bq/m^3 (US Environmental Protection Agency, 1982). In Sweden, concrete made from alum shales and containing up to 2620 Bq/kg radium-226 (one to two orders of magnitude higher than in typical construction materials) contributed to higher than normal levels of radon in houses (370–780 Bq/m^3) (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982; Swedjemark *et al.*, 1987). In the USA during the 1970s, houses constructed on reclaimed phosphate mining had indoor radon levels of up to about 520 Bq/m^3 (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Subsequent investigations showed that very high radon levels could also be found in houses built on soils uncontaminated by mining or milling residues in the USA (2.6–107 Bq/m^3), Sweden (up to 10 000 Bq/m^3), the UK (7–13 Bq/m^3), Canada (up to 700 Bq/m^3) and in other countries (US Radiation Policy Council, 1980; United Nations Scientific Committee on the Effects of Atomic Radiation, 1982; Nero, 1983; Wilson, 1984; Nero *et al.*, 1986).

A correlation has been reported between the concentration of uranium or radium in the ground and indoor radon or radon decay product levels, but it may not be adequate to allow prediction of probable indoor radon levels in specific areas and houses. Several factors influence the entrance of radon into a building: (i) the radium content of the ground, (ii) the ease with which radon can be transported through the ground, (iii) the availability of entry points into the building and (iv) the presence of a differential air pressure (Tanner, 1986; Sextro *et al.*, 1987).

The top few metres of soil are usually the most important source of the radon that finds its way into a building, since this region is in closest proximity to the building's foundations. Radium situated further away can be a significant source if nearby rock is fractured or if the soil consists of very coarse sand or gravel. International exploration of areas of uranium-rich soils and rocks, in order to find regions of economically recoverable uranium, may be of value in helping to identify regions in which uranium, and perhaps radium-226, concentrations near the earth's surface are higher than typical environmental levels (Peake & Rush, 1987).

Radon has been measured in gas in 'normal' soils throughout the world by various investigators at concentrations of about 7000–220 000 Bq/m^3 (Sextro *et al.*, 1987). Since radon levels within most structures range from about 10 to 150 Bq/m^3, it is evident that, if only a small percentage of the air inside a building comes from the soil gas underneath it, the amount of radon in the building could be significant. Radon concentrations in soil gas of $>1 \times 10^6$ Bq/m^3 have been measured in Sweden (Wilson, 1984) and of $>35 \times 10^6$ Bq/m^3 in Texas, USA (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Elevated radon concentrations have been found in houses in areas with high levels of radon in soil gas in Pennsylvania, USA (Gunderson *et al.*, 1987). In areas where soil gas contains high quantities, the ground is generally the dominant source of radon.

The availability of radon corresponds to the inherent ability of the ground to supply radon to structures built on it. This is dependent principally upon (i) the concentration of radon in the soil at the point from which soil gas escapes, (ii) the rate of extraction of soil gas and (iii) the pressure difference needed to extract it. In Sweden, radon availability has been characterized by classifiying land into that with high, normal and low potential for supplying indoor radon, on the basis of the geological characteristics of an area and the potential radon concentration of soil gas that those characteristics might produce (Wilson, 1984). The criteria are listed in Table 1.

Table 1. Classification of land areas with respect to potential for supplying indoor radon[a]

Potential	Radon in soil gas (Bq/m^3)	Geological characteristics
High	>50 000	High radium-226 in soil (>125 Bq/kg); uranium-rich alum shales, granites, pegmatites, uranium mineralization; eskers and porous soils
Normal	10 000–50 000	Normal radium-226 in soils (35–125 Bg/kg); gneisses, volcanics, till and sand
Low	<10 000	Low radium-226 in soil (<35 Bq/kg); fine sand, silt, moist clay; soils impermeable to movement of soil gas

[a] From Wilson (1984)

Scandinavian granites have been found to correlate with high indoor radon levels, and in Sweden the presence of granites is a basis for classifying an area as having 'high' radon potential. In New England, USA, water in many wells drilled into granite contains very high radon concentrations (Hess *et al.*, 1980). Gneisses found in the eastern part of the USA are often enriched with uranium and are very permeable; in an area of Pennsylvania, known as the Reading Prong, such deposits have resulted in extremely high indoor radon levels (Gunderson *et al.*, 1987). Swedish alum shales and dark shales in New York, USA, contain high radium concentrations and also contribute to high indoor radon levels. In the marine environment, uranium was often precipitated with phosphate, and in phosphate mining areas of the USA high indoor radon levels have frequently been observed (Guimond *et al.*, 1979; Tanner, 1986).

In Scandinavia, long, sinuous ridges of stratified gravels and sands, associated with glaciation and known as eskers, are sources of elevated indoor radon levels because of their particularly high permeability (Wilson, 1984; Castrén *et al.*, 1987).

[The Working Group noted that, although in many surveys of indoor radon some correlation has been noted between indoor radon levels and the characteristics of the underlying soil, rock or topography, the actual radon concentrations in buildings are affected by many other factors and cannot be predicted accurately by geological factors alone.]

All building materials contain some radium-226, and the radium content of different building materials around the world varies widely. Some materials, such as wood and natural gypsum, contain very little radium; others, which are often by-products of chemical and mineral extraction processes, have much higher concentrations. Examples of the latter are phosphogypsum, which is produced in the manufacture of phosphoric acid; red mud bricks, which contain a waste-product from the production of alumina from bauxite; and phosphate slag, which is waste from the production of elemental phosphorus. Alum shales in Sweden have been used for several decades in the manufacture of aerated concrete and for some years they provided about one-third of the building materials in Sweden. Production was stopped in 1979. The lower radium content in aerated concrete manufactured from alum shale between 1974 and 1979 results from a reduced content of alum shale (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

The radium-226 content of a building material is not the only factor that determines its significance as a source of radon in a building. Exhalation of radon from a building material is also influenced by its porosity, by coatings, as well as other characteristics. For example, although phosphate slag contains radium concentrations substantially higher than normal, relatively small amounts of radon are released because it is a glass-like material that does not facilitate radon transport. Other materials like phosphogypsum do release substantial amounts of radon (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982; Bruno, 1983).

(*d*) *Occurrence in water supplies*

Surface-waters generally contain very low concentrations of radon. In a survey of 25 water systems in the USA, a population-weighted average of about 685 Bq/m^3 was found; in only two systems were there >3700 Bq/m^3 (Horton, 1985). Consequently, surface-water systems probably make a negligible contribution to radon levels in a building. However, ground-water supplies that are rich in radon can be a significant source. The results of various investigations suggest that most of the radon that enters a building from ground water is quickly desorbed through typical household uses of running water. For typical use patterns and an air exchange rate of about one per hour, the average air-to-water concentration ratio is about $0.4-1.5 \times 10^{-4}$ (Partridge *et al.*, 1979; United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). This means that household use of ground-water with a radon concentration of about 7000–20 000 Bq/m^3 typically contributes about 1 Bq/m^3 radon to the air within the house.

Radon concentrations in ground-water throughout the world vary widely. In the USA, extensive surveys of public water supplies indicate that most do not have large concentrations of radon (Hess *et al.*, 1985a; Horton, 1985). Table 2 lists data for private wells, public ground-water supplies and public surface-water supplies. Private wells generally contain much higher radon levels than public water systems; and the data suggest that the larger the public water supply system, the lower the radon concentration of the water. Individual private wells in Maine, USA, have been shown to contain up to about 37 × 10^6 Bq/m^3. About 80% of the nearly 60 000 public water supplies in the USA are from ground-water; more than 90% of ground-water supplies serve fewer than 3300 people (Hess *et al.*, 1985a), implying that large metropolitan areas are generally served by surface-fed systems.

Table 2. Radon-222 concentrations in water supplies in the USA (Bq/m^3)[a]

State	Private well	Public water supply[b]	Public ground-water supply	Public surface-water supply
AL	4 440 (22)	300 (31)	2 600 (182)	ND (8)
AR	8 510 (2)	51 800 (1)	440 (22)	ND (1)
AZ	–	–	9 250 (124)	ND (6)
CA	1 590 (6)	29 200 (2)	17 400 (15)	ND (2)
CO	–	–	8 500 (76)	–
DE	–	–	1 100 (72)	–
FL	222 000 (34)	11 800 (2)	1 100 (327)	–
GA	77 700 (2)	1 630 (32)	2 480 (225)	1 590 (2)
IA	–	–	8 140 (85)	–
ID	–	–	3 660 (155)	–
IL	–	–	3 520 (314)	–
IN	–	–	1 300 (185)	–
KS	–	–	4 440 (47)	2 740 (2)
KY	55 500 (10)	ND (18)	1 180 (104)	ND (5)
MA	37 000 (8)	260 (2)	18 500 (212)	1 410 (2)
ME	259 000 (24)	36 600 (71)	–	–
MN	51 800 (1)	22 200 (1)	4 810 (233)	–
MO	ND (2)	–	890 (138)	ND (2)
MS	–	9 620 (2)	850 (104)	–
MT	159 000 (8)	– 8 500	(71) ND	(6)
NC	560 (29)	1 000 (2)	2 920 (404)	ND (4)
ND	–	16 300 (2)	1 300 (133)	–
NH	51 800 (18)	330 (12)	34 800 (52)	ND (6)
NJ	–	–	11 100 (38)	–
NM	2 180 (14)	1 670 (8)	2 035 (171)	ND (18)
NV	–	–	7 030 (57)	–
NY	55 500 (4)	1 260 (20)	1 920 (292)	ND (1)
OH	–	–	2 920 (165)	–
OK	–	–	3 440 (83)	–

Table 2 (contd)

State	Private well	Public water supply[b]	Public ground-water supply	Public surface-water supply
OR	16 700 (18)	–	4 440 (69)	ND (4)
PA	33 700 (16)	–	14 060 (105)	–
RI	240 500 (69)	192 400 (6)	88 800 (575)	ND (10)
SC	40 700 (28)	–	4 810 (384)	ND (14)
SD	155 400 (2)	2 180 (2)	7 770 (155)	–
TN	ND (2)	ND (2)	440 (98)	–
UT	–	–	5 550 (195)	–
VA	20 700 (42)	–	12 950 (284)	ND (4)
VT	7 770 (23)	31 100 (4)	24 400 (71)	480 (16)
WI	27 000 (40)	1 040 (4)	5 550 (278)	ND (12)
WY	–	–	12 200 (32)	ND (2)
USA	34 000 [424]	2 500 (224)	4 800 [6332]	37 [127]

[a] From Hess *et al.* (1985a); geometric means with number of samples in parentheses

[b] May include both ground-water and surface-water supplies

ND, not detected above background levels; –, no data available

In Finland, ground-waters appear to contain relatively large amounts of radon, with many reported values of about 3.7 × 10^6 Bq/m^3. In Sweden, public ground-water contains far less radon, and about half of the population obtain water from surface supplies. In the UK, natural radon levels in water supplies are lower than those in many other countries (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Table 3 summarizes data on radon concentrations in water supplies in several countries.

Table 3. Radon concentrations in water supplies[a]

Location	Number of wells with radon concentration in water				Radon concentration in water (× 10^3 Bq/m^3)	
	<37 × 10^3 Bq/m^3	37–370 × 10^3 Bq/m^3	0.37–3.7 × 10^6 Bq/m^3	3.7–37 × 10^6 Bq/m^3	Maximum	Average
Austria					7	1.5
Salzburg						
Finland						
Helsinki and Vantaa	4	12	65	29		1 200
Other areas	11	34	30	7	45 000	280
Italy	41	16	2	–		80
Sweden	155	17	–	–	150	19
USA						
Aroostock, Maine	13	19			200	48
Cumberland, Maine	1	6	7	2	5 800	1 000
Hancock, Maine	1	3	11	1	4 600	1 400

Table 3 (contd)

Location	Number of wells with radon concentration in water				Radon concentration in water (× 10^3 Bq/m^3)	
	<37 × 10^3 Bq/m^3	37–370 × 10^3 Bq/m^3	0.37–3.7 × 10^6 Bq/m^3	3.7–37 × 10^6 Bq/m^3	Maximum	Average
USA (contd)						
Lincoln, Maine	3	6	10	1	1 600	560
Penobscut, Maine	–	10	6	–	2 400	540
Waldo, Maine	–	5	9	–	3 100	1 100
York, Maine	–	6	9	–	2 200	670
All seven counties, Maine	18	55	52	4	5 800	660
North Carolina	85	117	10	–	1 700	100

[a] From United Nations Scientific Committee on the Effects of Atomic Radiation (1982)

–, no data available

2.2 Exposure

(*a*) *Occupational exposure*

Workers can be exposed to radon in several occupations. In the past, some groups of underground uranium miners were exposed to high levels of radon and its decay products (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Further information on exposure levels reported in epidemiological studies is given in section 3.3.

Exposures to radon decay products in uranium mines in certain countries are shown in Table 4 (United Nations Scientific Committee on the Effects on Atomic Radiation, 1982; Kleff, 1987), and levels in some nonuranium mines are shown in Table 5. [The Working Group noted that, over the past few decades, improvements in ventilation and working conditions in uranium and other underground mines have reduced exposure to radon and its decay products.]

Other underground workers and certain mineral processing workers may also be exposed to significant levels. Occupational exposure may also arise due to employment in buildings in areas with high radon levels. Exhalation of radon from ordinary rock and soils and from radon-rich water can cause significant radon concentrations in tunnels, power stations, caves, public baths and spas. The levels of radon decay products in various nonmining occupational environments are listed in Table 6.

In most developed countries, action has been taken to establish limits for occupational exposure to radon decay products, particularly for underground miners. The International Commission on Radiological Protection (ICRP) (1981) has recommended an annual limit of 0.02 J for the potential α energy intake of short-lived radon decay products by inhalation.

Taking into account the breathing rate of workers, this intake limit corresponds to an annual limit of 4.8 WLM for exposure to radon decay products by inhalation. The ICRP stresses that this limit should be interpreted as the lower boundary of an unacceptable exposure region and that occupational exposure should be kept as low as reasonably achievable below this limit. Furthermore, the Commission stipulates that the limit be reduced if external and internal exposures to other occupational sources of radiation are of relevance at the work place.

Table 4. Concentrations of and exposure to radon decay products in uranium mines

Country	Year	Average potential α energy concentration (WL) [Bq/m^3 EEC_{Rn}]	Average annual α energy exposure (WLM)	No. of miners	No of miners with exposure >4 WLM[a]
Argentina[b]					
Underground	1977–79		2.4	286–379	
	1980		2.4	95	0
Open-pit	1980		0.12	285	0
Canada[b]					
One leaching	1978		0.38	630	
Four underground	1978		0.74	3 690	
One open-pit	1978		0.41	276	
	1978		0.72	4 535	9
	1979		0.74	6 883	1
France[b]	1971	0.18 [666]			
	1972	0.17 [629]			
	1973	0.18 [666]			
	1974	0.13 [481]			
	1975	0.11 [407]			
	1978		2.0	1 284	~140
	1979		1.4	1 503	51
Italy[b]	1975	<1 [<3 700]			
USA[c]	1974		1.14	2 464	15
	1975		1.07	3 344	47
	1976		0.99	4 306	4
	1977		0.91	5 315	11
	1978		0.92	6 679	40
	1979		0.60	14 598	73
	1980		0.51	13 282	13
	1981		0.64	7 399	<7
	1982		0.62	5 083	5
	1983		0.73	2 135	0
	1984		0.68	1 557	2
	1985		0.43	1 219	0

[a] The maximum permissible exposure in many countries

[b] From United Nations Scientific Committee on the Effects of Atomic Radiation (1982)

[c] From Kleff (1987)

Table 5. Concentrations of and exposure to radon decay products in nonuranium mines[a,b]

Country	Year	Average potential α energy concentration (WL) [Bq/m^3 EEC$_{Rn}$]	Average annual α energy exposure (WLM)	No. of miners	No of miners with exposure >4 WLM
Finland	1972–74	0.2–0.4 [740-1480]		1300/23	
	1975–77		0.38	1370/16	0
Italy	1975	0.01–0.6 [37–2222]		2500/16	~75
Norway	1972	0.07 [259]	0.64	1870/33	
	1980	0.05 [185]	0.45	1380/23	
Poland	1970				
Copper		1–2 [3700–7400]			
Iron		1 [3700]			
Pyrite		4 [14 800]			
Phosphate		0.8 [2960]			
Zinc and lead		0.9 [3330]			
Baryte		0.2 [740]			
Coal		0.1 [370]			
South Africa	1973		1.7	320 000	
Sweden	1970		4.8	4800/5	2000
	1974		2.1	4600/5	360
	1975		1.9	5300/45	270
	1976		1.7	5300/46	225
	1977		1.6	5200/45	475
	1978		0.9	5300/47	270
	1979		0.7	4400/35	0
	1980		0.7	4400/35	0
UK	1968	0.01[c] [37]		220 000/420	
	1976		2–3[d]	2000/80	560
National coal	1981		0.12	185 200	
Private coal	1981		0.24	1500	
Other than coal	1981		2.60	2346/108	94
USA	1975	0.31 [1147]			
	1976	0.22 [814]			
	1977	0.12 [444]		/163	

[a] If not otherwise stated, the mines are iron, zinc, lead, copper or gold mines.

[b] From United Nations Scientific Committee on the Effects of Atomic Radiation (1982)

[c] This value is considered 'typical' for large nationalized coal mines.

[d] Based on measurements in about 80% of all noncoal mines

Blank spaces, no data available

Table 6. Concentration of radon decay products in working places other than mines[a]

Country	Working place	Average potential α energy concentration (WL) [Bq/m^3]
Austria (Badgastein)	Public baths	0.5–0.9 [1850–3330]
Hungary	Three caves (guides)	0.45[b] [1665]
Italy	20 spas	0.001 [4]
Japan	Two caves	0.8[b] [2960]
Sweden	Tunnels for water and cables; defence installations; hydroelectric power stations	∼0.1–1 [370–3700]
USA	Six caves	0.3–1 [1110–3700]

[a] From United Nations Scientific Committee on the Effects of Atomic Radiation (1982)

[b] An equilibrium factor of 0.05 is assumed.

(*b*) *Domestic exposure*

The range of average radon levels measured indoors varies greatly, from about 3 Bq/m^3 to >160 Bq/m^3 worldwide (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). In outdoor air, average annual radon levels throughout the world vary much less (0.1–10 Bq/m^3). In coastal areas and over islands and oceans, levels are generally between 0.1 and 3 Bq/m^3, whereas over large continental masses concentrations of between 3 and 10 Bq/m^3 are common (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982). Table 7 gives values for indoor radon concentrations in various countries. These levels and the large fraction of time spent indoors make the indoor environment a principal factor in the total exposure of individuals to radon and its decay products.

Table 7. Radon levels in dwellings in various countries

Country	Type of survey	No. of houses surveyed	'Average' radon concentration (Bq/m^3)	Range (Bq/m^3) and (type of distribution)
Austria[a]	Regional	729	22 (mean)	max. 220 (log-normal)
Belgium[b]	Pilot survey	78	41 (median) 50 (mean)	10–263 (log-normal)
	National	300	NA	NA
Canada[b]	National	14 000	33 (mean)	(log-normal)
Denmark[b]	Various small surveys	450	50 (geom. mean)	5–700
	National	500	NA	NA
Finland[b]	National	2154	63 (median)	9.4% >800;
	SW region	754	370 (mean)	≤13 000

Table 7 (contd)

Country	Type of survey	No. of houses surveyed	'Average' radon concentration (Bq/m³)	Range (Bq/m³) and (type of distribution)
France[b]	National	1056	44 (median)	3–1258; 5% >200
Germany, Federal Republic of[b]	National	6000 (approx.)	40 (median) 49 (mean)	1% >200; approx. max. 2000
Greece[b]	Regional	37	20 (mean)	3–136
Ireland[b]	Pilot	278	37 (median)	3–1190 (log-normal)
	National	300	61 (median)	17–1740 (log-normal)
Italy[b]	Regional			
	Milan	261	56 (mean)	max. 132;
	Umbria	70		85–292
	National	~1000	25 (median)	5–154
Japan[c]	Regional	258	31.4 (arith. mean); 18.8 (geom. mean)	0.116–289
Luxembourg[b]	National	12	40 (mean)	6.5–78
Netherlands[b]	National	1020	24 (median)	8–118
Norway[d]	National	1500	160 (arith. mean)	30–5300
Sweden[b]	National	756	69 (geom. mean)	11–3300
	Various	32 548	not applicable	3348 houses >400
Switzerland[b]	National	123	60 (median)	15–4000
UK[b,e]	National	2240	22 (arith. mean); 14 (geom. mean)	0–1100 (log-normal)
USA	National[f]	1377	55 (arith. mean); 34 (geom. mean)	3% >300
	National[g]	10 251	157 (arith. mean); 64 (geom. mean)	10% >300
	Pacific Northwest[h]	20 203	41 (arith. mean)	4% >150

[a] From United Nations Scientific Committee on the Effects of Atomic Radiation (1982)

[b] From McLaughlin (1986)

[c] From Aoyama *et al.* (1987)

[d] From Stranden (1987)

[e] Cliff *et al.* (1987)

[f] From Nero *et al.* (1986)

[g] From Alter and Oswald (1987)

[h] From Bonneville Power Administration (1987)

geom. mean, geometric mean; arith. mean, arithmetic mean; NA, not available

[The Working Group noted that many different techniques have been used for assessing radon levels, making it difficult to compare results. In some surveys, residents volunteered for testing, while in others houses were chosen randomly; studies involving volunteers may be biased to overestimated exposure. Some surveys were based on grab samples, while others involved long-term measurements. Differences in the time of the year at which a survey is conducted might bias the estimate of the long-term average; for example, measurements made in winter are generally higher than those taken in summer.]

In Austria, radon and its decay products were measured over an extended period, to correct for time-dependent variations, in 729 homes in Salzburg. The radon concentrations were observed to be distributed log normally, and the mean was 22 Bq/m^3. A mean equilibrium factor of 0.56 was found. The maximal radon concentration observed was about 220 Bq/m^3 (United Nations Scientific Committee on the Effects of Atomic Radiation, 1982).

In Belgium, a pilot study involved 78 houses that were selected as being geographically representative of the country. The indoor radon concentrations were (Bq/m^3): median, 41; mean, 50; minimum, 10; and maximum, 263, as measured by α-track detectors (Karlsruhe type; see section 2.3). Although the number of houses in the survey was small, modern houses tended to have higher concentrations, possibly because of less ventilation or the presence of building materials with higher radon emanation rates. Higher values were found also in areas such as the Ardennes, where natural stone is used extensively as a building material (McLaughlin, 1986).

In Canada, the main survey of radon and radon decay products was conducted in the summer months (June to August) of 1977, 1978 and 1980 (McLaughlin, 1986). About 14 000 single-family houses were surveyed in 19 cities. Those in Vancouver, British Columbia, had the lowest mean value for radon (5.2 Bq/m^3), whereas those in Winnipeg, Manitoba, had the highest mean value (57 Bq/m^3). The mean radon concentration in the survey was 33 Bq/m^3. [The Working Group noted that, in this survey, grab samples were taken in each house during the summer, and the data probably do not represent long-term mean values. Indoor radon concentrations would be higher in winter than in summer because houses are more often closed, more radon enters because of the 'capping' effect of frozen ground, and negative pressures are created in the houses by chimney effects.]

In Denmark, approximately 400 houses were selected randomly for pilot surveys, and radon was measured by different techniques, although α-track dosimeters were used principally. The results of the radon measurements were (Bq/m^3): geometric mean, 50; minimum, 5; and maximum, 700 (Sørensen *et al.*, 1985; McLaughlin, 1986).

In Finland, about 2200 single-family houses in 108 different locations were surveyed in 1983 (McLaughlin, 1986) by placing α-track dosimeters for two months. Large differences in median radon levels were observed among regions: in the populated south-east, the median value was about 216 Bq/m^3, over 9.4% of houses having radon levels greater than 800 Bq/m^3; for the entire country, the median value was 63 Bq/m^3. The mean indoor radon concentration was higher than the previously estimated national concentration, 90 Bq/m^3, probably because the estimate was not based on random sampling of geographical regions (Castrén *et al.*, 1985, 1987).

In a region of Helsinki, Finland, the mean radon level in 754 houses was 370 Bq/m^3; the highest and lowest local area means within the region were 1200 and 95 Bq/m^3, respectively. Indoor radon concentrations in excess of 2000 and 800 Bq/m^3 were found in 32 and 90 houses, respectively. The highest seasonally-adjusted average radon concentration detected in one house was 13 000 Bq/m^3. The principal source of radon appeared to be from the soil under the structures. Although no simple relationship between soil type and indoor radon was found, a dependence on both soil permeability and uranium content was observed. In Finland, the mean indoor radon concentration in houses built on eskers was twice as high as that in houses on other types of ground (Castrén *et al.*, 1985).

In France, 1056 houses were chosen for radon measurements in an area covering approximately 40% of the nation and about 8.3 million dwellings. The houses were selected as representative of the country's housing stock, and the number in each area was chosen to reflect the population density. Radon measurements were obtained for one-month periods using α-track dosimeters (CEA type LR-115, open face). The lowest median value (19 Bq/m^3) occurred in the Maritime Alps and in Paris and its suburbs. The highest median value (116 Bq/m^3) was found in the Loire region. The national median value was 44 Bq/m^3 (95% confidence interval, 7–220 Bq/m^3) (McLaughlin, 1986).

In the Federal Republic of Germany, radon measurements, completed in 1984, were made in approximately 6000 dwellings chosen from among addresses supplied by local council and police authorities, representing approximately every 5000th dwelling in the country. α-Track dosimeters (Karlsruhe type) were exposed over a period of about three months in each dwelling: one was placed in a bedroom and another in the main living area. Radon concentrations were (Bq/m^3): median, 40; mean, 49; and maximum, 2000; approximately 1% exceeded 200. The regional distribution of indoor radon levels was similar to the regional distribution of the local dose rate from terrestrial components of natural radiation exposure. The mean radon level was found to be significantly higher in dwellings in Bavaria and the Rhineland-Palatinate than in other areas (Schmier & Wicke, 1985; McLaughlin, 1986).

In Ireland, a pilot study of indoor radon levels in dwellings took place between 1982 and 1984, in preparation for a more extensive survey, which is in progress (McAulay & McLaughlin, 1985; McLaughlin, 1986, 1987). In the completed pilot study, 278 dwellings were chosen in a quasi-random fashion in selected areas of the country where uraniferous deposits were known to exist. Because of the use of selected areas, the data are not representative of the entire country. The large survey covers about 2000 dwellings representative of the national housing stock, and data are available for about 220 houses. α-Track detectors were used in both the pilot study and the national survey; however, CR-39s were used for the pilot study and LR-115s for the national survey. The detectors were usually placed in a bedroom for a period of three to six months. The results of the pilot study were (Bq/m^3): mean, 37; minimum, 3; and maximum, 1189. The preliminary results of the national survey are (Bq/m^3): median, 61; minimum, 17; and maximum 1740. The pilot study focused mainly on areas where geological characteristics suggested that enhanced radon levels might be present; however, some of the highest levels were not found in those areas. For example, in the pilot study a clustering of high indoor radon concentrations was found in the Cork area, which was not initially believed to be a likely place for high levels. Subsequently, these levels were shown to be due to very localized, high emanation rates in the soil. [The Working Group noted that, since high radon emanation rates can often be a localized phenomenon, only limited conclusions can be drawn from measurements of a small number of dwellings. These data suggest that the geological nature of the soil is more important than are construction characteristics of the house in determining indoor radon levels.]

In Italy, three surveys of varying size and distribution have been conducted (Sciocchetti *et al.*, 1985; McLaughlin, 1986). A national survey was made of about 1000 houses throughout the country, which were chosen in a random manner; however the sampling may not be representative of the average Italian situation. A second survey was conducted in 1984 in 261 houses in the Milan area, which were primarily the homes of employees of the organization conducting the survey and were neither chosen randomly nor necessarily representative of the whole area. The third survey was conducted in 1985 in about 70 houses in the Orvieto-Umbria region of central Italy. The measurement technique used by all three groups was passive α-track detectors (CR-39; ENEA-Casaccia type). One detector was placed in the living room and one in a bedroom, and the detectors integrated exposure over three or four months. The results of the national survey showed that the distribution of radon in the bedrooms was log normal. The radon concentrations (Bq/m^3) were: nationally — median, 25; minimum, 5; maximum, 154; in the Milan survey — mean (136 houses), 56; maximum, 132; and in the Orvieto-Umbria survey — mean, 154; minimum, 85; maximum, 292. The relatively high values in Orvieto are associated primarily with the use of volcanic tuff as a building material; materials with specific activities of 300–700 Bq/kg radium-226 are used in various parts of Italy.

In Japan, data are available for 258 houses in five areas of the country: Hiroshima, Nagasaki, Mihama, Misasa and Hokkaido (Aoyama *et al.*, 1987). Radon measurements were made with passive α-track detectors (CR-39), which were usually placed in the living room and left in place for four to ten months; some measurements were made in bedrooms. Only seven of the houses were in the Hokkaido area, and different measurement methods were used. The results for Nagasaki, Hiroshima, Misasa and Mihama were (Bq/m^3): arithmetic mean, 31.4; geometric mean, 18.8; median, 17.7; minimum, 0.116; and maximum, 289. The highest geometric mean (32.4 Bq/m^3) was obtained in Mihama and the lowest (9.9 Bq/m3) in Nagasaki. The investigators considered that the differences could be explained partially by different geological formations: granite rock predominates in the Hiroshima, Misasa and Mihama areas, while igneous rock is more prevalent in the Nagasaki area. Very little difference was found with regard to construction materials.

In the Netherlands, a pilot study and a larger national survey of radon were conducted during the mid-1980s (Put *et al.*, 1985; McLaughlin, 1986). On the basis of volunteers and statistical information about the Dutch housing stock, a sample of 1000 houses was selected to be representative of the overall housing in the country. The radon measurements were made using passive α-track dosimeters (Karlsruhe type), which were placed in the living room and, in some houses, in other rooms or in the crawlspace. Measurements were made over long periods — generally up to one year. The results of the survey were (Bq/m^3):

median, 24; mean, 29; minimum, 8; and maximum, 118. There appeared to be only a small seasonal effect on indoor radon concentrations, the median for the summer being only about 10% lower than that for the winter/spring period. The authors reported that indoor radon concentrations appeared to be related to certain housing characteristics: in houses with good insulation, the radon concentration was generally higher, irrespective of the year of construction. For houses built since 1970, an additional increase in the average radon concentration was observed, due perhaps to a more air-tight building shell, which may reduce the air exchange rate. By a comparison of levels in rooms on different floors and in crawlspaces, it was concluded that radon from the soil entering a house *via* the crawlspace is one of the main sources of indoor radon.

In Norway, a national survey of indoor radon levels was conducted in 1500 houses in 75 municipalities. The radon measurements were made by the activated charcoal method and were taken during winter when home heating was in use. Each dosimeter was left in the house for five to seven days. The results of the survey were (Bq/m^3): arithmetic mean, 160; maximum, 5300. The highest values were found in areas of alum shale and granite; high values were also found in very porous (glacial eskers) ground. In alum shale areas, radon levels ranged from 30 to 5300 Bq/m^3; in granite areas, from 30 to 800 Bq/m^3; in eskers, from 100 to 3000 Bq/m^3; and in 'normal' soil, from 20 to 200 Bq/m^3. Concentrations of radon were also measured in ground-water and building materials: the contributions from the ground, from building materials and from well-water were estimated to be 10–5500, 5–50 and 0–1000 Bq/m^3, respectively (Stranden, 1987).

In Sweden, the occurrence of high indoor radon levels in dwellings has been known and investigated for a number of years (McLaughlin, 1986; Swedjemark *et al.*, 1987). The work of Hultqvist (1956) on indoor radiation indicated that radon decay products could cause high exposure in the lung. Average levels of radon at that time ranged from 20 to 69 Bq/m^3; those in the mid-1970s were 50–440 Bq/m^3 (Edling *et al.*, 1986). A nationwide survey was conducted in the early 1980s in which measurements were made using passive radon monitors containing calcium sulphate:dysprosium thermoluminescent dosimeters, with a two-week integrating period. Data were obtained from 506 dwellings — 315 detached houses and 191 apartments in multi-family houses. The results were (Bq/m^3): arithmetic and geometric means in detached houses, 122 and 69, respectively; arithmetic and geometric means in apartments, 85 and 53, respectively; minimum value, 11; and maximum value, 3300. The difference between the two categories of dwelling was considered to be due to the lower ventilation rates in detached houses and to their greater direct contact with the ground. One of the reasons that many houses in Sweden have high radon levels is that extensive areas of the country are uraniferous and consist of geological formations, such as granite, pegmatites and alum shales, with relatively high contents of radium-226. Several hundred thousands of houses in Sweden are constructed with alum shale (McLaughlin, 1986; Swedjemark *et al.*, 1987).

Measurements have also been carried out by local authorities in Sweden. In a survey of about 32 500 dwellings, completed in 1982, 3348 dwellings were found to have radon decay product (equilibrium equivalent) concentrations >400 Bq/m^3, and 105 had values >2000 Bq/m^3 (McLaughlin, 1986). A comparison of radon levels in houses built before 1946 and those built or modified in the 1960s or later indicated that the levels in the older houses (measurements made in 1955–56) were four times lower than those in newer houses (measurements made in 1980–82). The difference was considered to be due to several factors, including the greater use of alum shale, reduced air exchange rates, more openings to the ground in the foundations and greater negative pressures in the newer houses (Swedjemark *et al.*, 1987).

In Switzerland, a survey was carried out in 123 single-family houses during the winter of 1981–82. Measurements were made on the ground and first floors of the houses using passive radon dosimeters (Karlsruhe type). The minimum radon concentration was 15 Bq/m^3 and the maximum, 4000 Bq/m^3; the most frequent radon levels appeared to range from about 20 to 70 Bq/m^3. A further study was conducted in Switzerland during the winter of 1982–83 in 105 single-family houses in order to determine the influence of energy conservation practices on radon levels. Radon levels in energy-efficient houses were increased on average by a factor of about 1.8 over those in conventional houses. Studies in Switzerland have also suggested that local geology is the most important factor influencing the radon content of indoor air (McLaughlin, 1986).

In the UK, a national radon survey was conducted in the early 1980s (McLaughlin, 1986; Cliff *et al.*, 1987; National Radiological Protection Board, 1987) to measure radon concentrations in 2240 representative dwellings selected from among the approximately 20 million dwellings in the country. Radon measurements were made using α-track detectors (CR-39) over a period of one year in the living area and main bedroom of each house studied. The radon concentrations were (Bq/m^3): arithmetic mean, 25; geometric mean, 15; maximum, 11 000. The highest average levels and the maximal value occurred in Cornwall (Bq/m^3): arithmetic mean, 520; geometric mean, 210; maximum, 11 000, due to the presence of uraniferous shale in the ground. The lowest indoor radon levels were found in Manchester. Over 100 000 dwellings in the UK were estimated to have very high indoor radon levels.

In the USA, Nero *et al.* (1986) analysed data from 1377 houses in 38 areas not believed to have particularly high radon levels. The resulting values (log-normally distributed) were (Bq/m^3): arithmetic mean, 55; and geometric mean, 34. Levels >300 Bq/m^3 occurred in 1–3% of the houses. Since extremely high levels of radon were reported in Pennsylvania and New Jersey, large-scale monitoring of radon has been carried out by the general population. α-Track detectors and activated charcoal have been used widely. One supplier of α-track detectors has accumulated over 60 000 measurements throughout the country covering periods of two to three months. The maximum value found was about 160 000 Bq/m^3. In 19 states, there were houses with maximal radon concentrations >3000 Bq/m^3 (Alter & Oswald, 1987). [The Working Group considered that these measurements cannot be representative of the country.]

Results of radon surveys in Montana, Idaho, Washington and Oregon are summarized in Table 8. α-Track detectors were used for three months to one year. The largest percentage of high values was found in Montana and Idaho, although the highest individual values were found in Oregon and Washington (Bonneville Power Administration, 1987).

Table 8. US radon levels — Pacific Northwest region (Bq/m^3)[a]

State	No. of sites	Highest reading	Arithmetic mean
Oregon	6 480	2 490	45
Washington	13 106	3 420	33
Idaho	530	1 720	174
Montana	67	990	138

[a] From Bonneville Power Administration (1987)

2.3 Analysis

Methods for the measurements of radon and its decay products have been reviewed (Budnitz, 1974; Organisation for Economic Co-operation and Development, 1985; George, A.C., 1986).

(*a*) *Integrating methods*

α-Track detectors are devices consisting of a small piece of plastic, which can be encased in a container with a filter-covered opening or left bare. α Particles emitted by radon decay products in air strike the plastic and produce tracks of submicroscopic damage. At the end of the measurement period, the detectors are returned to a laboratory where the plastic is placed in a caustic solution that accentuates the damage tracks so that they can be counted under a microscope or by an automated counting system. The number of tracks per unit area is correlated with the radon concentration in air using a conversion factor derived from data generated at a calibration facility. Many factors contribute to variability in results obtained with α-track detectors, including differences in detector response between batches of plastic, nonuniform deposition of decay products inside the detector holder, differences in the number of tracks used as background, variations in etching conditions and differences in readout. The variability in results decreases with the number of net tracks counted, so that counting more tracks over a larger area of the detector reduces the uncertainty of the result. Various configurations have been used for measurements indoors (Alter & Fleischer, 1981). Recent advances in track-counting techniques that allow automatic counting over larger areas of the detector may permit measurement periods to be reduced to several weeks (Glenwood Laboratories, 1986). [The Working Group noted that the primary advantage of α-track detectors is that they can produce an integrated measurement of radon concentration over a 12-month period; they are therefore useful devices for estimating cumulative exposures.]

Electrostatic-thermoluminescence detectors operate on the principle of electrostatic collection of charged radon decay products (Khan & Phillips, 1985a; George, A.C., 1986). In one configuration, commonly known as the passive environmental radon monitor, radon diffuses through a desiccant, which reduces the effect of humidity on the electrostatic collection efficiency. A thermoluminescent dosimeter, consisting of either lithium fluoride (Nyberg & Bernhardt, 1983) or calcium fluoride:dysprosium (Schiager, 1974), detects the α activity collected on the electrode. A second thermoluminescent dosimeter is used for subtracting background γ exposure. The exposure period can range from one week to months if the desiccant is changed. The primary disadvantage of this method is its sensitivity to humidity (Khan & Phillips, 1985b). The reduction in collection efficiency with increasing humidity can be diminished, however, by replacing the desiccant frequently.

Integrating instruments measure concentrations of radon decay products indoors or in mines over periods of days to weeks. These instruments require power to move air through a filter which collects radon decay products. The α activity on the filter is detected by either a lithium fluoride thermoluminescent chip or an α-track detector (Schiager, 1974; Guggenheim *et al.*, 1979; Nyberg & Bernhardt, 1983). [The Working Group noted that the relative expense, the need for a power source and occasional filter saturation after measurement periods of more than seven days make use of these instruments incompatible with a measurement period of months or longer.]

Adsorption techniques involve the use of activated carbon to adsorb radon in ambient indoor air. After the radon has diffused passively into the exposed carbon bed, it is allowed to decay into γ-emitting decay products, which are subsequently analysed using sodium iodide or germanium (lithium) γ detectors. The amount of activated carbon used in these 'charcoal canisters' ranges from about 30 to 200 g, according to size (Cohen & Cohen, 1983; George, A.C., 1984; Pritchard & Mariën, 1985; George, A.C., 1986). The limit of detection depends on the amount of carbon used and is usually <1 pCi/l [<37 Bq/m^3] for a two-day exposure (George, 1984). One design incorporates a diffusion barrier covering the charcoal bed, which limits the rate at which radon can diffuse into the carbon. This effectively increases the time during which the charcoal adsorbs the radon and allows a longer measurement period (Cohen & Cohen, 1983). The adsorption efficiency is affected by both humidity and temperature, and the canister should be weighed before and after exposure to determine moisture gain in order to calculate adjustment factors. Alternatively, inclusion of a desiccant in the device can reduce moisture gain. [The Working Group noted that the primary advantages of the passive activated carbon adsorption technique are that it is inexpensive and simple and that devices can be mass produced and delivered by post. The disadvantage is that it cannot be used to measure radon concentrations over a period longer than the saturation point of the quantity of charcoal in the bed; this is usually seven days or less, which makes this method unsuitable for estimating long-term exposures.]

(*b*) *Continuous methods*

Several types of radon and radon decay product detectors produce results on a continuous or semicontinuous basis and allow concentrations to be tracked over a period of time ranging from several hours to days, or longer.

A widely-used type of monitor samples ambient air by pumping air into a scintillation cell after it has passed through a particulate filter to remove dust and radon decay products (Thomas & Countess, 1979; Nazaroff *et al.*, 1983). As the radon in the air decays, the ionized decay products attach to the interior surface of the scintillation cell. The radon decay products decay by α emission, and the α particles strike a zinc sulphide coating on the inside of the cell, causing scintillations, which are detected by a photomultiplier tube. In a second design, the air pump is eliminated and air is allowed to enter the device only by molecular diffusion; it thus requires a larger sensitive volume (Chittaporn *et al.*, 1981). The limit of detection of these methods ranges from 0.4 to 37 Bq/m^3 and depends on the size of the scintillation cell (George, A.C., 1986). [The Working Group noted that, since these instruments are relatively expensive and must be operated by trained personnel, they are more useful for intensive investigations at one location than surveys of many houses.]

Continuous radon decay product monitors can be used to sample ambient air by filtering airborne particles as the air is drawn through a filter cartridge at a low flow rate. An α detector, such as a diffused-junction or surface-barrier detector, counts the α particles produced by the radon decay products as they decay on the filter. [The detector is normally regulated to detect α particles with energies between 2 and 8 MeV.] The α particles emitted from the radon decay products polonium-218 and polonium-214 are the most significant contributors to the events that are measured by the detector (Thomas & Countess, 1979). The event count is directly proportional to the number of α particles emitted by the radon decay products on the filter. The limit of detection for such continuous monitors ranges from 0.001 to 0.01 WL, depending on the flow rate (George, A.C., 1986). [The Working Group noted that these instruments are very costly and their use is limited to special studies.]

(*c*) *Grab sampling methods*

Grab sampling methods involve very short-term (minutes) measurements of radon or radon decay product concentration. Scintillation cells, which have been used extensively for this purpose, have a transparent window with a zinc sulphide coating on the interior. The window is placed on a photomultiplier tube, and the scintillations that result from α disintegrations produced by interaction with the zinc sulphide are counted. The cell can be filled either by prior evacuation or by using a portable pump. The analysis is performed about 4 h after filling to allow the short-lived radon decay products to reach equilibrium with the radon. Samples of air can also be collected in metal containers or bags that are impermeable to radon for subsequent transfer to scintillation cells. The limit of detection for grab sampling of radon is dependent on the volume of the cell used, but can reach 0.1 pCi/l [3.7 Bq/m^3] for a 65-l tube (George, A.C., 1986; George, J.L., 1986).

The concentration of radon decay products can be measured by collecting the decay products from a known volume of air on a filter and counting the α activity on the filter during or following collection. The Kusnetz procedure has been used extensively in measurements in mines to assess the total concentration of radon decay products (Kusnetz, 1956). In this procedure, decay products from up to 100 l of air are collected on a filter during a 5-min sampling period. The total α activity on the filter is counted at any time between 40 and 90 min after the end of sampling. The analysis can be carried out using a scintillation-type counter to obtain a gross α-count rate, which is converted to radon decay product concentration using the appropriate counter efficiency. Other counting intervals may also be used (Rolle, 1972).

The Tsivoglou procedure, as modified by Thomas, can be used to determine the concentrations of individual decay products (Thomas, 1972). Sampling is carried out in the same way as in the Kusnetz procedure; however, the filter is counted three times following collection: between 2 and 5 min, 6 and 20 min, and 21 and 30 min. Count results are used in a series of equations to calculate the concentrations of three decay products: polonium-218, bismuth-214 and lead-214.

Radon decay product activity on the filter can also be analysed by α spectrometry (Martz *et al.*, 1969; Tremblay *et al.*, 1979). Since this method requires the use of a smaller volume of air than the scintillation method, it is less sensitive and is used primarily for laboratory applications.

Errors associated with grab sampling measurements of radon decay product concentrations arise from a number of factors, including inaccurate timing, unstable air pumps, improper calibration of flow meters and leaks in filters and filter holders (Loysen, 1969).

[The Working Group noted that, since indoor radon concentrations vary considerably over time, the results of short-term measurements must be used with caution when estimating exposures. As indoor radon concentrations tend to follow a seasonal cycle, measurements over a 12-month period are most useful for estimating long-term exposures. In addition, concentrations at different locations in the same house often vary by a factor of two or more, concentrations generally being greater at lower levels (George, 1984; Hess *et al*, 1985b). Therefore, measurements that are used to estimate exposures must be evaluated critically with respect to both time period and location.]