

IONIZING RADIATION: SOURCES AND BIOLOGICAL EFFECTS

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on the Effects of Atomic Radiation

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NOTE

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ANNEX D

Exposures to radon and thoron and their decay products

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Introduction

1. Radon and thoron are among the first of the natural radioactive nuclides which were discovered, identified and examined at the beginning of this century. They are noble gases and are produced in nature by the decay of the radium isotopes ^{226}Ra and ^{224}Ra , respectively. Radium and its progeny have been

used for medical and technical purposes and for research for more than 70 years.

2. The interest of the Committee in radon and thoron lies in their detrimental effect on man. Since they occur in nature, man has always been exposed, mainly through inhalation of their decay products, to radon and thoron daughters. The detrimental effect has been

recognized since the sixteenth century, when miners in central Europe suffered from what was then called "Schneebergkrankheit". Later experience of mining of uranium in the middle of this century identified radon daughters to be a source of cancer.

3. Man is everywhere exposed to radon and thoron products. The major source of exposure consists of radon and its decay products, occurring in domestic housing. Radon emanates from building materials and from the ground below. In some countries the radiation dose to man caused by inhaled radon daughters constitutes more than 50% of the total radiation dose to man from other natural or artificial sources. Radon is also a major problem in the nuclear fuel cycle because it gives rise to an occupational exposure during uranium mining and is a source of exposure to the public during the storage of waste (tailings) from the milling of uranium ore. Other sources of radon include the burning of coal, emissions from geothermal plants, and radon dissolved in water.

4. There are no epidemiological data available about the effects of radon and thoron daughters on the public. However, the detrimental effect of radon and its decay products in indoor air can be estimated by extrapolation from information obtained from the exposure of uranium miners. Although the estimates of exposure in this case are far from perfect, they clearly demonstrate that high levels of radon and its daughters in air warrant close attention. The Committee wishes to draw particular attention to the fact that in a number of countries special efforts are now taken to conserve energy by insulation of houses and by decreasing the ventilation rate. It has been shown that such actions cause increased concentrations of radon and radon daughters in houses.

5. The purpose of this Annex is to provide information about the levels and doses of radon and thoron and their decay products and about physical parameters influencing and causing these levels and doses. The detrimental effects of radon and thoron daughters are not dealt with in this Annex.

I. GENERAL CONCEPTS AND PROPERTIES

A. PHYSICAL AND CHEMICAL PROPERTIES OF RADON AND THORON

6. Radon (^{222}Rn) and thoron (^{220}Rn) are naturally-occurring radioactive gases. Radon is produced by the decay of ^{226}Ra in the uranium series and decays by alpha particle emission to a polonium isotope (^{218}Po) which by further decay through isotopes of lead, bismuth, and polonium ends with a stable isotope of lead (^{206}Pb). The half-lives and radiation energies are shown in Table 1. Other physical data on radon are presented in Table 2. Thoron is produced by the decay of ^{224}Ra in the thorium series and decays by alpha particle emission to a polonium isotope (^{216}Po) which decays through isotopes of thallium, lead, bismuth and polonium to a stable isotope of lead (^{208}Pb), as shown in Table 1.

7. Radon and thoron are chemically inert, noble gases. They occur in almost all materials and for the most part (90% and more) are trapped in the solids carrying their precursors ^{226}Ra and ^{224}Ra . Accordingly, the measurement of the gamma radiation emitted from

the materials provides an indication of its uranium and thorium content as most of the gamma radiation from the ^{238}U and ^{232}Th series arises from the radon and thoron progeny.

8. Some of the radon and thoron may diffuse into other media such as the surrounding water or air and frequently the concentration of radon and thoron in surrounding water and air is higher than the concentration of ^{226}Ra and ^{224}Ra . Occasionally there may be a deficiency of radon in water relative to ^{226}Ra if the water is in open contact with air so that the radon may diffuse into air.

9. Because of the short half-life of thoron its concentration in water is generally insignificant, unless the water has a high concentration of ^{224}Ra . Radon can be transported a long way from its precursor and often there is no simple correlation between the radon concentration in air and water and the concentration of ^{226}Ra in adjacent materials. However, radon concentrations in air in uranium mines and near mill tailings are usually well correlated to adjacent sources.

B. MEASUREMENT AND IDENTIFICATION OF RADON AND THORON AND THEIR DECAY PRODUCTS

10. Radon and thoron are identified by their radiation or their daughter products. In measurements of very low activity concentrations in air (less than a few Bq m^{-3}) radon and thoron can be concentrated or trapped, using activated charcoal filters at normal or low temperature. At higher concentrations they can be directly collected and measured by ionization or scintillation methods [1]. They can also be measured continuously by passing air through the measuring equipment. In general the activity of radon and thoron in a sample is monitored by measuring the number of emitted alpha particles both from radon (thoron) and their progeny which build up in the collected sample. Radon in water is measured by gamma spectrometric methods directly on the sample (including liquid scintillation methods) or by measuring the radon released by bubbling a gas through the sample. The activity or potential alpha energy concentration (see paragraph 13) of daughters in air can be determined by drawing air through a filter and measuring the alpha particle activity with a ZnS or silicon surface barrier detector, thermoluminescent dosimetry or track etch detector. Because of the short half-lives of the radon daughters the measurement must be made simultaneously with the collection or shortly thereafter. Lead-212 and its decay products are measured some hours after collection following the decay of the short-lived radon daughters.

11. After decay of the short-lived radon daughters the long-lived isotopes of lead, bismuth and polonium complete the uranium series. The practical use of these isotopes as indicators of the current activity of radon and short-lived daughter products is limited. Even if the main part of ^{210}Pb in outdoor air comes from the decay of radon daughters in air, there is no simple correlation between ^{210}Pb and the local concentration of radon in air. Similarly, ^{210}Pb in water is not a reliable indicator of earlier radon levels in the water because of variable and unknown releases of radon from water to air during its passage from the radium source to the water outlet, of insufficient time to allow significant decay of radon in the water, of leaching of ^{210}Pb from the rock and finally because of depletion of the daughters by

adsorption on the rock surfaces. However, ^{210}Pb in the human body (in bone) has been used as an indicator for earlier inhalation of high concentrations of radon in mines [C8, H26].

C. SPECIAL QUANTITIES AND UNITS

12. The potential alpha energy $E_{\text{pot,at}}$ of an atom in the decay chain of radon or thoron is the total alpha energy emitted during the decay of this atom to ^{210}Pb or ^{208}Pb , respectively. The potential alpha energy per unit of activity (Bq) of a radionuclide j is $E_{\text{pot,at}}/\lambda_j$ where λ_j is the decay constant. Values of $E_{\text{pot,at}}$ and $E_{\text{pot,at}}/\lambda_j$ are listed in Table 3.

13. The potential alpha energy concentration of any mixture of (short-lived) radon or thoron daughters in air is the sum of the potential alpha energy of all daughter atoms present per unit volume of air. The usual unit for this quantity is MeV l^{-1} . This unit is related to the SI units J and m^3 according to $1 \text{ J m}^{-3} = 6.24 \cdot 10^9 \text{ MeV l}^{-1}$. A special unit for this quantity used for radiation protection purposes is the working level (WL). A WL is defined as a potential alpha energy concentration of $1.3 \cdot 10^5 \text{ MeV l}^{-1}$ of air. One WL corresponds approximately to the potential alpha energy concentration of short-lived radon daughters in air which are in radioactive equilibrium with a radon concentration of 3.7 kBq m^{-3} . For thoron daughters in radioactive equilibrium with thoron, one WL corresponds to a thoron concentration of 275 Bq m^{-3} . Quotients of potential alpha energy concentration (in WL) to activity concentration (in Bq m^{-3}) are listed in Table 3 for the individual isotopes. The conversions between the units used are as follows:

$$\begin{aligned} 1 \text{ J} &= 6.24 \cdot 10^{12} \text{ MeV} \\ 1 \text{ MeV} &= 1.6 \cdot 10^{-13} \text{ J} \\ 1 \text{ WL} &= 1.3 \cdot 10^8 \text{ MeV m}^{-3} = 2.08 \cdot 10^{-5} \text{ J m}^{-3}. \end{aligned}$$

14. The equilibrium equivalent concentration of radon or thoron, $\chi_{\text{eq,Rn}}$ and $\chi_{\text{eq,Th}}$, respectively, corresponding to a non-equilibrium mixture of short-lived radon or thoron daughters in air, is that activity concentration of radon or thoron in radioactive equilibrium with its short-lived daughters which has the same potential alpha energy concentration C_{pot} as the actual non-equilibrium mixture. The potential alpha energy concentration of radon and thoron daughters in equilibrium with 1 Bq m^{-3} of radon and thoron is given in Table 3. Accordingly, if the potential alpha energy concentration is C_{pot} , the corresponding equilibrium equivalent concentration for radon and thoron is obtained as indicated in Table 4.

15. The equilibrium factor F is defined as the ratio of the total potential alpha energy for the actual daughter concentrations to the total potential alpha energy of the daughters which would be in equilibrium with the radon or thoron concentration. Accordingly, F is calculated from the ratio of equilibrium equivalent concentration of radon or thoron to the actual radon or thoron concentration in air

$$F = \frac{\chi_{\text{eq,Rn}}}{\chi_{\text{a,Rn}}} \text{ for radon} \quad (1a)$$

and

$$F = \frac{\chi_{\text{eq,Th}}}{\chi_{\text{a,Th}}} \text{ for thoron} \quad (1b)$$

Sometimes the equilibrium factor for radon is assumed to be known, and the value of $\chi_{\text{eq,Rn}}$ is calculated by multiplying the measured radon concentration $\chi_{\text{a,Rn}}$ by F .

16. The activity exposure of an individual to radon or thoron is the time integral of the activity concentration of radon or thoron, respectively, to which the individual is exposed over a given time. The unit used here is Bq h m^{-3} . The exposure to radon or thoron daughters is expressed as "potential alpha energy exposure" or "radon or thoron daughter exposure", and is the time integral of the concentration (C_{pot} or $\chi_{\text{eq,Rn}}$ or $\chi_{\text{eq,Th}}$) of the daughter mixture to which the individual is exposed over a given time. Potential alpha energy exposure is expressed in the units:

$$\begin{aligned} 1 \text{ J h m}^{-3} &= 6.24 \cdot 10^{12} \text{ MeV h m}^{-3} = 4.8 \cdot 10^4 \text{ WL h} \\ 1 \text{ WL h} &= 1.3 \cdot 10^8 \text{ MeV h m}^{-3} = 2.08 \cdot 10^{-5} \text{ J h m}^{-3} \end{aligned}$$

Radon or thoron daughter exposure using the quantity equilibrium equivalent concentration of radon or thoron, respectively, is expressed in unit Bq h m^{-3} . The potential alpha energy exposure of miners is often expressed in the unit Working Level Month (WLM). One WLM corresponds to an exposure to a concentration of 1 WL for the reference period of 170 hours.

$$\begin{aligned} 1 \text{ WLM} &= 170 \text{ WL h} = 2.2 \cdot 10^{10} \text{ MeV h m}^{-3} = \\ &= 3.5 \cdot 10^{-3} \text{ J h m}^{-3} \\ 1 \text{ J h m}^{-3} &= 285 \text{ WLM} \end{aligned}$$

One WL corresponds to an equilibrium equivalent concentration of radon ($\chi_{\text{eq,Rn}}$) of 3700 Bq m^{-3} and a potential alpha energy exposure of 1 WLM radon daughter corresponds to a radon daughter exposure of $6.3 \cdot 10^5 \text{ Bq h m}^{-3}$. For thoron daughters 1 WL corresponds to a $\chi_{\text{eq,Th}}$ value of 275 Bq m^{-3} and 1 WLM corresponds to $4.7 \cdot 10^4 \text{ Bq h m}^{-3}$.

D. BUILD-UP AND EQUILIBRIUM RATIOS FOR RADON AND THORON DAUGHTERS

17. The build-up of radon or thoron activity is given by the expression

$$A(t) = F_r A_0 (1 - e^{-\lambda t}) \quad (2)$$

where $A(t)$ is the activity of radon or thoron contained in a confined unventilated space at time t (at time $t=0$ the activity is assumed to be zero), F_r is the emanating power which is the fraction of radon or thoron released into that space from the ^{226}Ra or ^{224}Ra source of activity A_0 , and λ is the decay constant for radon or thoron. The activity concentration in a confined unventilated space increases according to the same formula divided by the volume V of the space. If the space is ventilated with ventilation rate λ_v the increase of radon or thoron concentration follows the expression

$$\chi_a(t) = \frac{F_r A_0}{V \left(1 + \frac{\lambda_v}{\lambda}\right)} (1 - e^{-(\lambda + \lambda_v)t}) \quad (3)$$

where $\chi_a(t)$ is the activity concentration of radon or thoron in air, and λ_v is the ventilation rate defined as the number of air changes per unit time (h^{-1}), which is the quotient of the air flow rate ($\text{m}^3 \text{ h}^{-1}$) through the space and its volume (m^3). The build-up for radon is shown in Figure 1 for several values of λ_v . Because of the short half-life of thoron the equilibrium level of

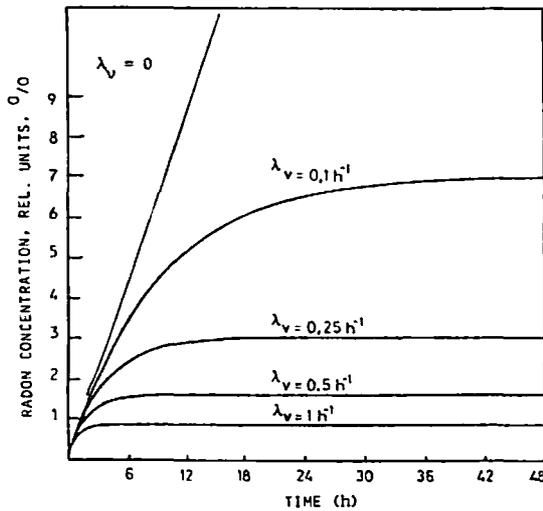


Figure I. The relative growth of radon concentration in a confined space as a percentage of the equilibrium value for no ventilation versus time for different ventilation rates λ_v (h^{-1}) [W14]

thoron is reached within 5 minutes and in practice is independent of the ventilation rate (assuming $\lambda_v < 10 \text{ h}^{-1}$).

18. Radon and thoron emanating into the air are practically free of decay products so that the daughter activity in air builds up with time. The build-up of daughter activity concentration in a volume of air containing a given radon or thoron activity concentration (the value of which is only affected by decay) is given by the following general expression

$$\chi_{a,j}(t) = \chi_a(0) \sum_{i=0}^j G_i e^{-\lambda_i t}$$

where

$$G_i = \frac{\prod_{k=1}^j \lambda_k}{\prod_{n=0}^j (\lambda_n - \lambda_i)} \quad (n \neq i) \quad (4)$$

$\chi_a(0)$ is the radon or thoron activity concentration at time $t=0$, and λ represents a decay constant. Values of $j=1,2,3$, etc. stand for the first, second, third etc. radon or thoron daughter, while $i=0$ represents the parent radon or thoron. By multiplying the relative activity of the daughters by their potential alpha energy, the corresponding growth of the potential alpha energy of the radon daughters can be estimated. This is shown in Figure II for a constant radon concentration. The increase of the potential alpha energy concentration C_{pot} during the first 40 minutes is given approximately by [E6]

$$C_{\text{pot}}(t) = C_{\text{pot}}(t_1) \left(\frac{t}{t_1} \right)^{0.85} \quad (5)$$

where t_1 is one minute.

19. It can be seen from equation (3) that the activity concentration of radon decreases as the ventilation rate increases, while the concentration of thoron is approximately independent of the ventilation rate (assuming $\lambda_v < 10 \text{ h}^{-1}$). The corresponding decrease of the activity concentration of the daughters can be estimated from the expression

$$\frac{\chi_{a,j}}{\chi_a} = \frac{j}{n-1} \frac{1}{1 + \frac{\lambda_v}{\lambda_n}} \quad (6)$$

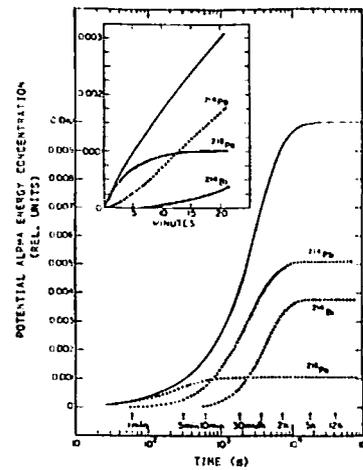


Figure II. Potential alpha energy concentration for an initially pure source of ^{222}Rn and the relative contributions to it from the progeny, as a function of time [C17]

where $\chi_{a,j}$ is the activity concentration of daughter j in air and χ_a is the activity concentration of radon or thoron in air. It is assumed that factors influencing the concentration of radon or thoron daughters in air other than radioactive decay and ventilation rate can be disregarded and that there is a constant emission of radon or thoron into the ventilated space. The corresponding decrease of potential alpha energy concentration is estimated by multiplying the activity concentration of each daughter by the respective potential alpha energy as given in Table 3 and taking the sum of these products.

1. Attachment of radon and thoron daughters to aerosols

20. At the time of formation of the first daughter, ^{218}Po or ^{216}Po is an unattached, predominantly (more than 90% [D5]) positive small ion or neutral atom (size of the order of $10^{-3} \mu\text{m}$). Some ions may be attached to molecules of water vapour and other gases. After a time of the order of 10–100 s the daughters will attach to an aerosol particle (normal size in the range 10^{-2} – $1 \mu\text{m}$). In the subsequent alpha decays the radioactive atom or ion may again become unattached because of the recoil energy when it decays. For example, the recoil energy in the decay of ^{218}Po (117 keV) is transferred to its daughter ^{214}Pb . Lead-214 will desorb, and appear as an unattached ion or neutral atom for a short time after its formation, but then becomes attached. This recoil effect has been found to apply to about 80% of the decays of ^{218}Po [M13]. The first decay product of thoron is ^{216}Po , which is very short-lived ($T_{1/2} = 0.15 \text{ s}$) and therefore unattached. The next decay product, ^{212}Pb , has a relatively long half-life ($T_{1/2} = 10.64 \text{ h}$) and therefore in room air it is predominantly attached.

21. Changes in the air concentration of the various radon and thoron daughters are governed by radon and thoron exhalation rate, radioactive decay, attachment and recombination, deposition and sedimentation, and transport by diffusion and air flow.

22. The attachment of radon or thoron daughters to aerosols depends on the diffusion coefficient of free ions and atoms and on the concentration and particle size distribution of the aerosol. The attachment to aerosols has been studied by several investigators [L6,

B2, B11, D8, K5, R2, W5]. The attachment rate can be expressed by

$$\lambda_a = \chi \int_0^{\infty} K(\Delta, r) n(r) dr \quad (7)$$

where χ is the number of particles per unit volume of air; Δ is the diffusion coefficient of free atoms in air; $n(r)dr$ is the relative number of particles in an increment (dr) of radius at the radius r ; $K(\Delta, r) = 4\pi r^2 \Delta h(1+hr)$; $h = \bar{v}/4\Delta$; \bar{v} is the mean gas kinetic velocity of free daughter atoms ($= 1.7 \cdot 10^4 \text{ cm s}^{-1}$ at 20°C) [J1].

23. If one assumes that the aerosol distribution indoors and outdoors is log-normal with a Count Median Aerodynamic Diameter (CMAD) of $0.05 \mu\text{m}$ and a geometric standard deviation of 3, and if one adopts representative values of 10^4 cm^{-3} for particle density and $0.054 \text{ cm}^2 \text{ s}^{-1}$ for Δ , then the calculated attachment rate λ_a is about 10^{-2} s^{-1} , corresponding to a mean life of the unattached radioactive ion of about 100 s. For working rooms or rooms with smokers, the conditions will be different. In a mine, for example, with $\text{CMAD} = 0.1 \mu\text{m}$ and $\chi = 10^5 \text{ cm}^{-3}$, the attachment rate will be 0.28 s^{-1} and a corresponding mean life of only about 4 s. The mean lives of free ions and the correlation between the concentration of free ions and aerosols have been studied repeatedly [D8, K5, R3].

24. The deposition of attached daughters to surfaces is determined by the diffusion of their carrier aerosols and by the quotient of the surface area S to the volume V of the room containing the aerosols. The deposition rate λ_d is given by the expression

$$\lambda_d = v_d \frac{S}{V} \quad (8)$$

where v_d is the deposition velocity (m s^{-1}). The value of v_d for the attached daughters depends on the size distribution of the carrier aerosols and normally lies within the range of 10^{-5} to 10^{-4} m s^{-1} . For unattached daughters the value of v_d lies in the range of 10^{-3} to 10^{-2} m s^{-1} [J1, A9]. For particles with normal Activity Median Aerodynamic Diameter (AMAD), i.e., $0.05\text{--}0.2 \mu\text{m}$, a deposition velocity of 10^{-4} m s^{-1} and S/V quotient of about 2 m^{-1} (which is a "normal" value for rooms), the deposition rate is about 0.7 h^{-1} [J1] and the mean life is accordingly about 1.4 h. For unattached daughters the deposition rate is of the order of 1 min^{-1} .

25. The diffusion coefficient of the unattached decay products of radon and thoron in air is of the order of $0.05 \text{ cm}^2 \text{ s}^{-1}$. Various values have been used within the range of $0.034\text{--}0.06 \text{ cm}^2 \text{ s}^{-1}$ [C5, D6, K11]. Decreasing diffusion coefficient with increasing humidity has been reported [R1] and may be due to clustering of the daughters or clustering of water molecules on daughter atoms [K11] in the air, which is not absolutely free of condensation nuclei (size in the range $0.01\text{--}0.1 \mu\text{m}$). Theoretical studies on the mechanism of the interaction of radon daughters and water vapour are presented in [H23]. If the concentration of condensation nuclei is very low and constant but the relative humidity changes from 5 to 90%, the diffusion coefficient for unattached thoron daughters is found to remain fairly constant ($0.05\text{--}0.06 \text{ cm}^2 \text{ s}^{-1}$) [K10]. The diffusion coefficient is also independent of thoron concentration in the reported range of $70\text{--}4000 \text{ kBq m}^{-3}$. However, if the concentration of condensation nuclei increases several orders of magnitude (e.g., by decreasing the ventilation rate in a room) the diffusion coefficient decreases to

about $0.007 \text{ cm}^2 \text{ s}^{-1}$ (at a concentration of $5 \cdot 10^4 \text{ cm}^{-3}$). These experiments were made in an aerosol-free atmosphere. In other experiments [P10], the diffusion coefficient of positively charged decay products of thoron has been measured under very dry conditions (relative humidity 2%) to be $0.048 \pm 0.003 \text{ cm}^2 \text{ s}^{-1}$ and with high relative humidity (30–90%) to be $0.068 \pm 0.004 \text{ cm}^2 \text{ s}^{-1}$. For neutral ^{212}Pb atoms the diffusion coefficient was $0.067 \pm 0.004 \text{ cm}^2 \text{ s}^{-1}$, irrespective of the humidity.

26. A study concerning the effects of charge, humidity and the number of condensation nuclei on radon daughter concentration in air has been made by Cooper et al. [C10]. The study laid particular emphasis upon uranium mines and showed great variations of the aerosol attachment and surface attachment (plateout) rates. The effect of humidity on the diffusion coefficient of radon daughters is dubious, although there seems to be agreement that the plateout effect strongly depends on the fraction of daughters which are not attached to particles [C9, M11]. The effect of turbulent air in mine environments has been found to lead to a reduction of the airborne radon daughters [W13, S9] owing to an increased plateout on the walls or plateout on the fan blades. The fraction of radon daughters which attaches to the blades of a fan that circulates the air is substantial if the relative humidity is low (5%) and negligible when the relative humidity is high (80%) [H14]. The effects under conditions of low humidity are illustrated in Figure III. Similar effects were also

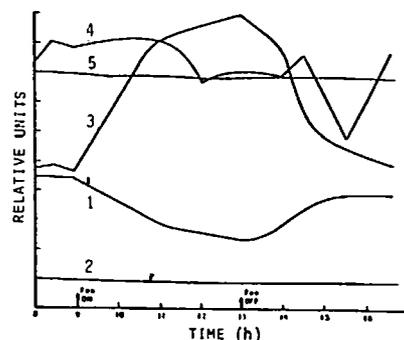


Figure III. Radon daughter concentration in the air (curve 1, full scale 10 WL), on the walls (curve 2, full scale 1000 counts/mln), and on the fan (curve 3, full scale 3000 counts/30 min) in response to the switching on and off the fan. Curve 4 (full scale 10^5 cm^{-3}), the condensation nuclei concentration, and curve 5 (full scale 70 kBq m^{-3}), the radon concentration, are practically independent of the fan being on or off. Humidity was $< 5\%$ [H14]

studied in room air [J12] and from preliminary results it was concluded that the removal by a fan system mainly affected ^{218}Po . In experiments with monodisperse aerosols (diameters $2\text{--}10 \mu\text{m}$) and radon and radon daughters the attachment rate of ^{218}Po was studied [H11]. It was concluded that the attachment rate was proportional to particle concentration and particle cross-section area, which implies that the attachment process of ^{218}Po to aerosols can be described by gas kinetic theory. The attachment rate was also found to be proportional to the radon or ^{218}Po concentration.

27. The equation governing the fraction of unattached radon or thoron daughters in a room involves terms for radioactive decay, attachment, deposition of aerosols and ventilation. Assuming that the outdoor concen-

tration can be neglected, the fraction of unattached ^{218}Po and ^{212}Pb atoms is given by

$$F_{rd} = \frac{1}{1 + \frac{\lambda_a}{\lambda_1 + \lambda_v + \lambda_d}} \quad (9)$$

where λ_a is the rate of attachment of free daughter atoms to aerosol particles; λ_1 is the decay constant; λ_v is the ventilation rate; λ_d is the wall deposition rate for attached daughter atoms [J1]; and F_{rd} is the quotient of the number of unattached ^{218}Po or ^{212}Pb atoms to the total number of ^{218}Po or ^{212}Pb atoms, respectively. Sometimes a fraction F'_{rd} is given which is the ratio of the number of unattached ^{218}Po or ^{212}Pb atoms to the number of ^{218}Po or ^{212}Pb atoms assumed to be in equilibrium with radon or thoron, respectively. This fraction corresponds to $F_{rd} F_1$, where F_1 is the activity equilibrium ratio of ^{218}Po or ^{212}Pb to radon or thoron, respectively; i.e., $F'_{rd} = F_{rd} F_1$.

28. With the typical value of λ_d given above (paragraph 24), it is obvious that for daughters with a high decay constant like ^{218}Po , λ_1 will be the most significant term in the above equation (9) for a given value of λ_a , while the ventilation rate λ_v within a normal range ($0-1 \text{ h}^{-1}$ in buildings) will be less significant. However, in mines with high ventilation rates the value of λ_v will significantly influence the value of F_{rd} for radon daughters. As increased ventilation may cause a decrease of the aerosol concentration, the value of λ_a will decrease, with an increase in the fraction of unattached radon daughters.

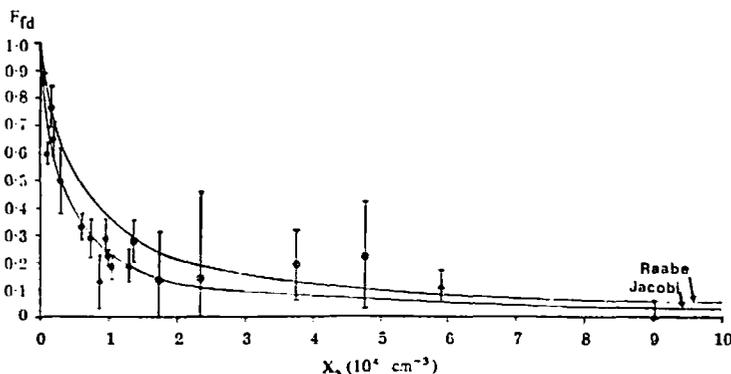


Figure IV. Variation of the fraction of unattached ^{218}Po as a function of the concentration of condensation nuclei. \blacktriangle — Tln mine; \blacksquare — Colley Hill (rural), \odot — Sutton (suburban); \bullet — artificial atmosphere [D8]

31. Activity concentrations of attached and unattached radon decay products in mines have been measured [R3] and the fraction F_{rd} (unattached activity/total activity of respective daughter) has been calculated for ^{218}Po , ^{214}Pb and ^{214}Bi [M12]. In 60 samples, mean and standard deviation for F_{rd} were obtained of $(8 \pm 10)\%$ (range 0.3–55.4%) for ^{218}Po , $(4.9 \pm 6.8)\%$ (range 0.06–31.4%) for ^{214}Pb and $(2.7 \pm 3.8)\%$ (range 0.08–16.4%) for ^{214}Bi .

32. In summary, the rate of attachment of radon and thoron daughters to aerosols normally lies in the range $10^{-1}-10^{-2} \text{ s}^{-1}$, with higher values in dusty air. The rate of deposition (plateout) on indoor surfaces is about 1 h^{-1} for attached daughters, and 1 min^{-1} for unattached daughters. Low humidity and increased air turbulence increase the plateout. The unattached fraction of radon and thoron daughters is normally in the range 1–10%. The unattached fraction increases with decreasing aerosol concentration and with increasing ventilation rate.

29. Measured values of F_{rd} and F_1 in thorium plants [M16] are $F_{rd} = (0.74 \pm 0.02)\%$ and $F_1 = (3.7 \pm 2.5)\%$ ($= ^{212}\text{Pb}/\text{Tn}$), giving $F'_{rd} = 0.027 \pm 0.021$ [K11]. Several values of the fraction of unattached radon daughters have been reported. In Sutton, in the United Kingdom, F_{rd} for ^{218}Po varied between 0.07 and 0.40 (relative to ^{218}Po with no consistent difference indoors and outdoors [D7]). In New York City, F'_{rd} (unattached fraction relative to radon) varied in two series between 0.01 and 0.06 (average 0.04) and 0.05–0.12 (average 0.09) [F3]. In Sterling Forest, F_{rd} for ^{218}Po was measured to be 0.04–0.1 (average 0.08) [G4] with the lowest value occurring on a rainy day. F'_{rd} for ^{218}Po indoors in New York has been measured to be 0.04–0.07 [F3, G4].

30. The fraction F_{rd} of unattached ^{218}Po atoms has been measured [D8] in open air and in mines as a function of the concentration X_a of condensation nuclei in the air. The measured values were compared with the theoretical expression

$$F_{rd} = \frac{\lambda_1}{\lambda_1 + k X_a} \quad (10)$$

where λ_1 is the decay constant for ^{218}Po and $k X_a = \lambda_a$. The values of k have been calculated by Jacobi and by Raabe to be $6 \cdot 10^{-5} \text{ min}^{-1} \text{ cm}^3$ and $4 \cdot 10^{-5} \text{ min}^{-1} \text{ cm}^3$, respectively [D8]. The comparison with experimental values is presented in Figure IV [D8].

2. Equilibrium factors for radon and thoron daughters in ventilated confined spaces

33. The daughter ratios given by equation (6) may be combined with the conversion coefficients of Table 3 to obtain the equilibrium factor F as a function of ventilation rate. These are presented in Table 5. Corresponding values for the potential alpha energy concentration of the daughters can be derived by multiplying F by the potential alpha energy concentration of the daughters if they were in equilibrium with the parent radon or thoron activity concentration χ_a . The potential alpha energy concentration for radon daughters calculated in this way is shown in Figure V as a function of ventilation rate [P12]. This calculation assumes no plateout, and negligible daughter concentration in the inlet air. For thoron daughters, the corresponding quantity simply follows the equilibrium factor F , as the thoron activity is fairly constant.

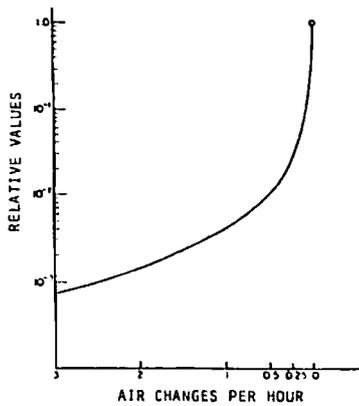


Figure V. Potential alpha energy concentration of radon daughters as a function of ventilation rate [P12]

34. When the effects of plateout and radon daughters in the inlet air are included, the simple description given above no longer applies. The degree of plateout to the walls is strongly influenced by the unattached fraction, which in turn affects the activity ratios and the equilibrium factor F . The equilibrium factor for radon daughters has been estimated from measurements made under varying aerosol conditions [W14] and the results are shown in Figure VI. It is apparent that average

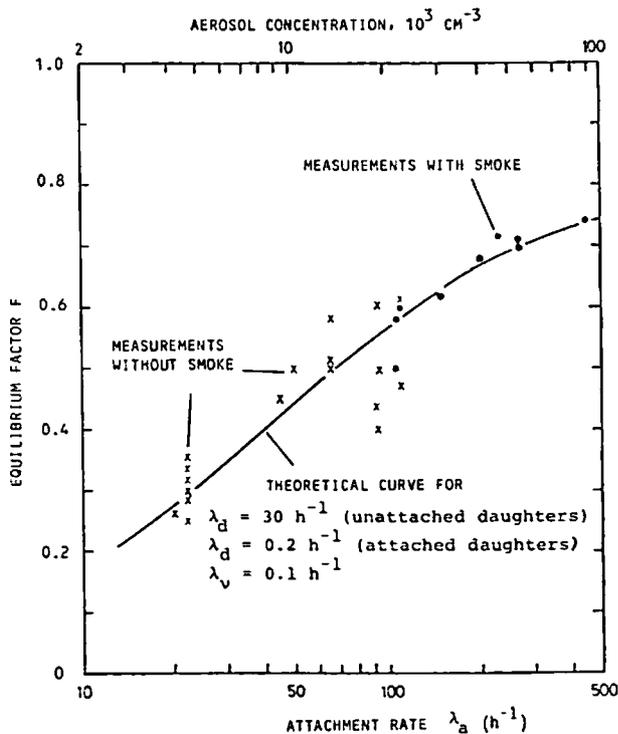


Figure VI. Equilibrium factor for radon daughters in a room as a function of the aerosol concentration and the attachment rate λ_a . The emanation rate into the room is $7.4 \text{ Bq m}^{-3} \text{ h}^{-1}$ [W14]

values of F corresponding to different values of the rate of attachment λ_a agree reasonably with the theoretical curve which includes the effect of plateout. Significant concentrations of radon daughters in the inlet air complicate the situation further. Figure VII illustrates the variation in the equilibrium factor F with ventilation rate for a number of exhalation coefficients R_v ($\text{Bq m}^{-3} \text{ s}^{-1}$) defined as the exhalation rate R ($\text{Bq m}^{-2} \text{ s}^{-1}$), multiplied by the quotient S/V (m^{-1}) of surface area to volume. For this calculation an equivalent equilibrium concentration of 2.6 Bq m^{-3} was assumed

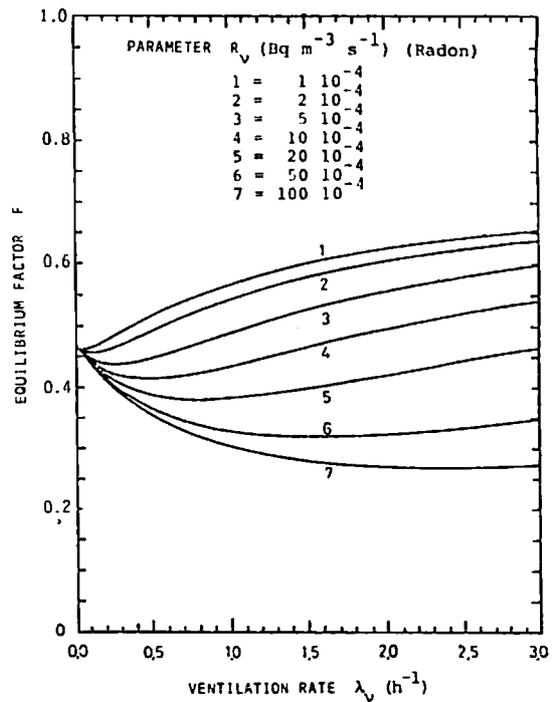


Figure VII. Equilibrium factor for radon daughters in a room as a function of ventilation rate λ_v with varying emanation rates into a room and the radon daughter concentration in outdoor air taken into account [W14]

for outdoor air, with an equilibrium factor of 0.75. In other theoretical calculations [J1] the equilibrium factor has been estimated to be as low as 0.2 for conditions of low aerosol concentration ($\lambda_a = 100 \text{ h}^{-1}$) and a ventilation rate of $\lambda_v = 1 \text{ h}^{-1}$. However, when allowance is made for the equilibrium factor for inlet air drawn from outdoors, the equilibrium factor increases to greater than 0.3 (calculated from [P9] for the reference house described in paragraph 105). For rooms with high aerosol concentrations ($\lambda_a > 1000 \text{ h}^{-1}$) the equilibrium factor F will be larger (with $\lambda_v = 1 \text{ h}^{-1}$, $F = 0.35$, according to [J1]). These results suggest that approximate estimates of F , which take plateout into account may be obtained by multiplying the values of F in Table 5 by 0.4 ($\lambda_a = 100 \text{ h}^{-1}$) or 0.6 ($\lambda_a = 1000 \text{ h}^{-1}$) for $\lambda_v < 2 \text{ h}^{-1}$ or by 0.5 if λ_a is unknown.

35. The effect of deposition on the walls (plateout) in radon daughter ratios in uranium mine atmospheres has recently been studied [H21]. It was concluded that selective plateout, particularly of ^{218}Po , is the main reason why it is impossible to find simple correlations between activity or potential energy concentrations and ventilation rate or age of the air.

36. Plateout also influences the factor F for thoron daughters. In rooms with low aerosol concentration and a ventilation rate of about 1 h^{-1} , F is about 0.009, instead of 0.058 as given in Table 5. In rooms with higher aerosol concentration F is about 0.013 [J1]. Other calculations show somewhat higher values [W14]. It seems appropriate therefore to assume that the equilibrium factor is generally less than half of the values given in Table 5.

37. If there is a ventilation system with recirculation of part of the air and also air filtration, there will be a decrease in the radon concentration proportional to the fraction of fresh air admitted (if the radon concentration in fresh air is neglected). However, if the air exchange rate, including recirculation, is unchanged,

the equilibrium factor F will not change, unless there is a decrease in the aerosol concentration because of recirculation.

II. RADON AND THORON IN OUTDOOR AIR AND IN WATER

38. The concentrations of radon and thoron and of their daughters vary substantially with time and space. In general, the activity levels in air make a much more significant contribution to radiation exposure than those in water. The levels of radon and thoron in air or water depend on four factors. These are the nature of the source, its emanating power, transport of radon and thoron from the source and their ultimate dispersal. These factors will be considered in this chapter.

A. SOURCES

39. Radium and thorium in soil are the main sources of radon and thoron in the global atmosphere. Both the radium and thorium concentrations usually lie in the range of $10\text{--}50\text{ Bq kg}^{-1}$ (Annex B). The average world-wide concentration of both elements in soil is assumed to be 25 Bq kg^{-1} (Annex B). In seawater the concentration of ^{226}Ra is 4 to 5 orders of magnitude lower [B25], and the concentration of ^{228}Th is 6 to 7 orders of magnitude lower [M27]. Nevertheless, for reasons which are discussed in paragraph 62, the exhalation rate from the oceans is only about 2 orders of magnitude lower than that for soil. Some published values of radon exhalation rates per unit area are listed in Table 6 [W15]. The estimated area-weighted average radon exhalation rate per unit area from soil, excluding Antarctica and the Greenland ice cap, is $16\text{ mBq m}^{-2}\text{ s}^{-1}$ [W15] corresponding to a total exhalation rate of $1.9\text{ }10^{12}\text{ Bq s}^{-1}$. The values reported in Table 6 have a log-normal distribution with a geometric mean of about two-thirds of the arithmetic mean. In a study on the flow of ^{210}Pb into the global atmosphere good agreement has been achieved between calculations based on the geometric mean of the radon exhalation rates per unit area and those based on glacier ice concentrations of ^{210}Pb [J16]. Other estimates of world-wide average soil radon exhalation rates per unit area are $15\text{ mBq m}^{-2}\text{ s}^{-1}$ [17] and $19\text{ mBq m}^{-2}\text{ s}^{-1}$ [B13]. Reported ranges are $0.2\text{--}70\text{ mBq m}^{-2}\text{ s}^{-1}$ [W15, G17]. The total annual radon exhalation is then $5\text{--}10\text{ }10^{19}\text{ Bq}$, corresponding to a total equilibrium activity of the order of 10^{18} Bq . As may be seen in Table 23, the concentration of ^{222}Rn in ground level air varies widely with geographical location, and is up to two orders of magnitude higher over land masses than in maritime locations. Other sources of radon in air are plants and ground water ($< 10^{19}\text{ Bq a}^{-1}$), natural gas (about 10^{14} Bq a^{-1}) and the combustion of coal (about 10^{13} Bq a^{-1}) [H7]. Radon from houses may contribute about 10^{16} Bq a^{-1} (assuming 10^9 reference houses of the type presented in Table 13, the true value may be between $5\text{ }10^{15}\text{--}10^{17}\text{ Bq a}^{-1}$). The total annual radon release from uranium mines, mills and tailings is of the order of 10^{15} Bq (assuming 120 GW(e) installed nuclear capacity and a normalized radon release of $2\text{ }10^{13}\text{ Bq [GW(e) a]}^{-1}$) (see Table 2 of Annex F).

40. Geothermal power stations are a source of radon which is minor from a global perspective but may be of local significance. At present only a small part of the world energy requirement is produced from geothermal sources (0.1%) [U5] but increases in the future are

possible. The production of hot steam in deep geological formations leads to releases of radon from water in the bed rock. Measurements performed in Italy [M4] indicate a radon release per unit energy generated of about $4\text{ }10^{14}\text{ Bq (GW a)}^{-1}$, which is three to four orders of magnitude higher than the normalized release from coal-fired plants. The radon concentration in geothermal steam at Wairakei and Broadland in New Zealand is $100\text{--}400\text{ Bq kg}^{-1}$ steam which results in an annual radon release of 10^{11} to 10^{12} Bq .

41. Another source of interest is phosphate mining. Rock phosphate is used mainly as a source of phosphorus for fertilizers. Morocco, the Soviet Union and the United States are the main producers. The United States produces 40% of the total world production of phosphate rock and more than 80% of the total United States production occurs in Central Florida. Sedimentary phosphate ores such as those in Morocco and Florida often have a higher concentration of uranium and its decay products than magmatic ores such as those from Kola. Most phosphate is mined in an open-pit or strip-mining process. In order to reach the phosphate rock, the overburden and a second layer called the leach zone have to be removed. The leach zone has generally a higher concentration of ^{226}Ra than the overburden and is afterwards frequently placed at or near the surface, resulting in increased ^{226}Ra concentrations [C17]. The subsequent radon exhalation rate from phosphate regions depends on land reclamation practices. A quantitative characterization is given in Figure VIII, which refers to phosphate regions in Florida [R8].

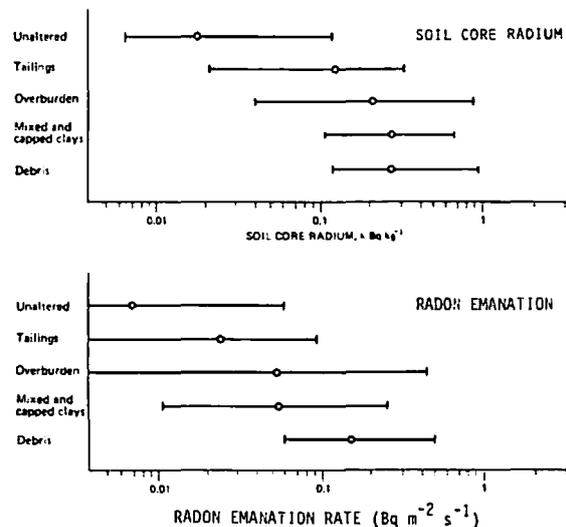


Figure VIII. Radiological characteristics of Polk County lands, Florida (geometric means and ranges by land type) [R8]

42. Volcanic activity as a source of radon in air has been investigated in Iceland, Japan and Hawaii [W16, B14, 18]. Radon concentrations in the volcanic plumes were observed to be above ambient air concentrations and in Hawaii concentrations of the order of 50 Bq m^{-3} were reported. The radon release rate was estimated to be of the order of 10 GBq h^{-1} which is negligible compared with the radon release rate from the island surface. The radon release from the eruption of Mount St. Helens in the United States on 19 May 1980 has been estimated to be about 10^{17} Bq of radon in the plume [F6], which corresponds to about 10% of the total atmospheric inventory. However, this estimate has not been confirmed by other observers.

43. Radium-224 in soil is the main source of outdoor thoron. However, because of the relatively short half-life of ^{224}Ra (3.7 d) it must be continuously supported by its precursors. Thus, in practice, ^{228}Th ($T_{1/2}$ 1.9 a) and ^{228}Ra ($T_{1/2}$ 5.8 a) control the thoron exhalation rate. Even though the activity exhalation rate for thoron is much higher (about 100 times) [B13, H7] than that for radon, the total global activity inventory in the atmosphere is much less (about 100 times less) than that for radon, primarily because of the shorter half-life of thoron.

B. MECHANISMS OF RADON AND THORON RELEASE

44. The physical process of the radon and thoron release from the structure of the rocks is not yet fully understood. Only a part of the radon and thoron atoms produced by decay in a particle is released to the surrounding water or air. The fraction of the radon or thoron atoms formed in a solid which makes its way into the pores of the medium, and thereby becomes amenable to transport, is defined as the emanating power of the solid for radon or thoron. Some mechanisms describing the release of radon are reviewed in [A2, T6].

45. The transport and release of radon and thoron from a solid into air or water occurs through diffusion and flow of the air or water. High porosity increases the diffusion rate. In a very dry solid the release of radon or thoron is reduced by readsorption of radon and thoron atoms on surfaces in the pores and fractures in the solid. If the solid is slightly moist, the radon and thoron release is enhanced up to a certain moisture content, above which it decreases again because of the lower diffusion rate in water-filled pores [M10].

46. The exhalation of radon per unit area for large particles and pieces of uranium ore has been demonstrated to increase with the size of the particles [T4]. This effect may be due to the presence of microfracturing in the samples. Similarly, the exhalation of radon from ordinary rock samples may be less than the exhalation from rock walls in a mine, because of greater fracturing and the presence of cracks in the walls caused by blasting [L9, K7]. Table 7 shows the results of some measurements of radon exhalation from core material and a surface in a mine [L9].

C. MECHANISMS OF RADON AND THORON TRANSPORT IN THE GROUND

47. Once radon and thoron have entered the surrounding water or air phase, they are further transported by diffusion, mechanical and convective flow, and by percolation. The diffusion process can be expressed by the formula

$$\chi(x) = \chi(0) \exp\left(-x / \sqrt{\frac{\Delta_k}{\lambda}}\right) \quad (11)$$

where $\chi(x)$ is the radon or thoron concentration at distance x in water or air from the surface; $\chi(0)$ is the radon or thoron concentration at the surface; Δ_k is the effective diffusion coefficient and λ is the decay constant [G8]. The diffusion coefficient Δ_k for radon is about $0.1 \text{ cm}^2 \text{ s}^{-1}$ in air and $10^{-5} \text{ cm}^2 \text{ s}^{-1}$ in surface water [T1]. In soil Δ_k is usually of the order of $10^{-2} \text{ cm}^2 \text{ s}^{-1}$ or

less [T1]. This means that radon will have decayed to 10% of its original value after a diffusion through 5 m of air, 5 cm of water and about 2 m in soil. Migration of radon for much longer distances by diffusion is not important. Accordingly, the transport of radon over greater distances in water and air must occur through other mechanisms. The short half-life of thoron (55 s) limits its migration to less than one per cent of that of radon [J2]. Therefore, the remainder of this discussion relates mainly to radon.

48. Radon in rock or soil may also be transported by mechanical forces in the earth causing changes of pore space through compressive stresses. These may be caused by earth tides or by intermittent forces such as those of earthquakes. Such mechanisms may explain intermittent variations in radon exhalation from soil surfaces.

49. Another possible transport mechanism is thermally-induced fluid convection. Some data suggest that this process may be important, but further results are needed [M14].

50. In the main, however, the transport of radon through soil and bedrock by water probably depends on the percolation of water through the pores and along fracture planes of the bedrock. The radon concentration in water from a percolation zone $\chi_{w,Rn}$ is related to the radium concentration of the percolated rock $C_{\text{rock,Ra}}$, the rock density, ρ_{rock} , the emanating power, F_r , the fractional pore space of the rock (porosity), $F_{\text{rock,ps}}$, and the water velocity v , through the formula

$$\chi_{w,Rn}(x) = \frac{F_r \rho_{\text{rock}} C_{\text{rock,Ra}}}{F_{\text{rock,ps}}} (1 - e^{-\lambda x/v}) \quad (12)$$

where λ is the decay constant and x is the distance which water travels through the rock [A2]. It follows from this formula that heavy rainfall causes a decrease in radon concentration as the velocity v increases. Thus, if the water in a well is supplied by water from a percolation zone of higher than normal radium content, the radon concentration in the well is likely to decrease after rainfall. However, if the percolation water is a minor source for the well, heavy rainfall might flush out radon-rich water from the percolation zone thereby causing an increase of radon concentration. Variations of radon concentration in well water are sometimes observed to be seasonal [A2, K8, S11].

51. The radon content in ground water and water in springs and wells is seen from equation (12) to depend on the radium concentration in the rock of the aquifer. Consequently, the highest radon concentrations are found in water in the vicinity of uranium ore bodies. High concentrations ($> 400 \text{ kBq m}^{-3}$) are also found in wells bored in granite, pegmatite, syenite and porphyry. Low radon concentrations occur in ground water from basic rocks and sedimentary rocks, such as limestone and sandstone [K8].

52. In investigations of radon in ground water in six counties in Maine, United States, the highest concentrations were found in granites and adjacent sedimentary rocks which have experienced great metamorphic change. Low concentrations were found in ground water from non-granitic areas with little metamorphic change. The average radon concentration in ground water in granite zones was about 1 MBq m^{-3} and in chlorite zones 40 kBq m^{-3} [H10].

53. The radon concentration in water may vary with the depth of the wells if the size of the percolation zone depends on the well depth. This effect has been found to be significant for wells in granite up to depths of around 50 m, as can be seen in Figure IX. At depths

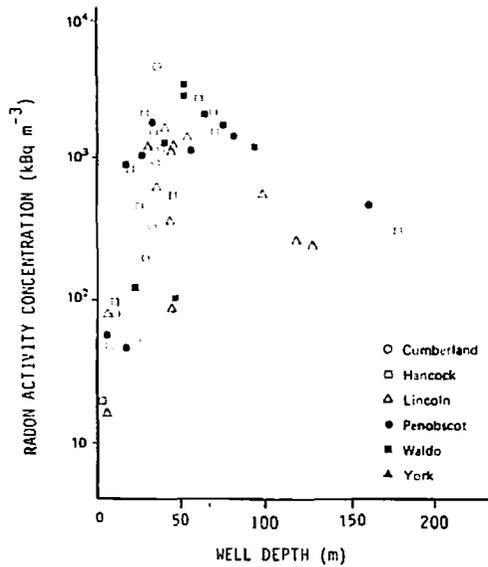


Figure IX. The variation of ^{222}Rn concentration as a function of water well depth in granites [H10]

beyond 100 m there was a tendency for radon concentrations to decrease [H10]. In other investigations no significant correlation of radon concentration with depth has been observed [S11].

54. Measurements of radon in soils can be used in uranium exploration, particularly when the uranium ore is covered by a thick layer of soil (more than about 0.5 m) [D9], although anomalies arising from selective radium or radon transport from the uranium ore may give rise to errors in interpretation. Meteorological factors, particularly those affecting the moisture content of the soil, can result in large variations in radon concentration in the soil [U6, C1].

55. Radon measurements in sub-surface waters and in dry boreholes have recently been used as an earthquake predictor [S10, S37, S38, K16, N1]. It is postulated that dilation of the rock caused by strain releases radon to the ground water and subsequently increases the radon content in well and spring waters in the vicinity. This method is however subject to many complicating factors and there is still considerable doubt about its practical applicability.

D. RADON AND THORON TRANSFER THROUGH SOIL AND EMANATION TO AIR

56. Radon and thoron enter the air contained in soil by diffusion from soil particles or sometimes from radon-rich ground water at greater depths. The concentration of radon and thoron in this air decreases with decreasing distance from the surface because the gases escape to the open air above the ground. As the exhalation increases, the gas concentration in soil air decreases, and vice-versa.

57. The diffusion mechanisms for radon and thoron are the same. For radon, the diffusion through soil to surface can be expressed by the diffusion equation [J2]

$$\frac{dX_{a,Rn}(z)}{dt} = \Delta_k \frac{d^2X_{a,Rn}(z)}{dz^2} + \frac{F_r \lambda_{Ra} X_{soil,Ra}}{F_{soil,ps}} - \lambda_{Rn} X_{a,Rn}(z) \quad (13)$$

where $X_{a,Rn}(z)$ is the concentration (per unit volume of soil air) of radon atoms at depth z ($z=0$ at the ground surface); Δ_k is the effective diffusion coefficient ($\text{cm}^2 \text{s}^{-1}$); F_r is the emanating power ($0 < F_r < 1$); λ_{Ra} and λ_{Rn} are the decay constants of radium and radon, respectively; $X_{soil,Ra}$ is the concentration (per unit volume of soil) of radium atoms; and $F_{soil,ps}$ is the porosity of the soil. Under steady state conditions $dX_{a,Rn}(z)/dt = 0$, and the concentration of radon activity at depth z is given by

$$[\chi_{a,Rn}(z)]_{soil} = \frac{F_r \chi_{soil,Ra}}{F_{soil,ps}} \left[1 - \exp\left(-z \sqrt{\frac{\lambda_{Rn}}{\Delta_k}}\right) \right] \quad (14)$$

where $[\chi_{a,Rn}(z)]_{soil}$ is the radon activity concentration per unit volume of soil air at depth z ; $\chi_{soil,Ra}$ is the radium activity concentration per unit volume of soil.

58. At great depths ($z \rightarrow \infty$) the equilibrium radon concentration in soil air is

$$\frac{F_r \chi_{soil,Ra}}{F_{soil,ps}}$$

For an infinitely thick soil, the exhalation rate is given by

$$R = \left(\Delta_{eff} \frac{d[\chi_{a,Rn}(z)]_{soil}}{dz} \right)_{z=0} \quad (15)$$

where $\Delta_{eff} (= F_{soil,ps} \Delta_k)$ is the effective bulk diffusion coefficient. The combination of equations (14) and (15) gives

$$R = \lambda_{Rn} F_r \chi_{soil,Ra} \sqrt{\Delta_k / \lambda_{Rn}} \quad (16)$$

The rate of diffusion is highly dependent on the moisture content of the soil and the diffusion coefficient (Δ_{eff}) can vary by several orders of magnitude. A value of $5 \cdot 10^{-2} \text{ cm}^2 \text{ s}^{-1}$ for Δ_k has been suggested as typical for soil [S39]. Assuming that $F_r = 0.1$, $\chi_{soil,Ra} = 0.074 \text{ Bq cm}^{-3}$ and $\Delta_k = 0.05 \text{ cm}^2 \text{ s}^{-1}$,

$$R = 2.4 \cdot 10^{-2} \text{ Bq m}^{-2} \text{ s}^{-1}$$

which compares reasonably with measured average values of $1.5\text{--}2.0 \cdot 10^{-2} \text{ Bq m}^{-2} \text{ s}^{-1}$. The corresponding value for thoron is $1.9 \text{ Bq m}^{-2} \text{ s}^{-1}$ if all constants except λ are the same. Several studies of thoron exhalation have been reported [D4; G16, M1, M10, S40] and the average observed value of the thoron exhalation rate from soil is around $1 \text{ Bq m}^{-2} \text{ s}^{-1}$ [J2].

59. The concentration of radon and thoron in soil air is affected by meteorological factors each as barometric pressure, humidity, rainfall and temperature. Rising barometric pressure has been found to increase the radon concentration in soil air whereas falling pressure causes a decrease of the radon concentration [B15]. However, a tendency for decreasing concentration with increasing atmospheric pressure has also been observed for radon in soil air. This decrease is believed to be due

to a flow of air from outside into the soil surface layers forcing the soil gas initially in surface layer to greater depths. The results of some recent studies are shown in Tables 8 and 9 [19], which show the influence of depth, barometric pressure and wind speed on the concentration of radon and thoron in soil air. The standard deviation of the observations are 50% for radon and 25% for thoron. The effect of wind speed is believed to be due to increased turbulence at the ground surface which causes a pumping effect on the soil gases.

60. Factors influencing the concentration in soil air also influence the exhalation rate, but in the opposite direction. Rain, snow, freezing and increased atmospheric pressure reduce the exhalation rate, while higher wind speeds and temperature increase it. Consequently, the radon concentration in soil has its maximum values in winter, when the ground is frozen, and in rainy periods, unless the rain flushes out the radon from the soil. Radon exhalation rates vary for different soils. In areas with a high radium burden, such as uranium mining areas, the exhalation rate is much higher than normal (more than two orders of magnitude greater for uncovered tailings) [U9]. Diurnal variations of exhalation rate have been reported for radon [M2], while others have found the exhalation rate to be relatively constant between day and night [W15, G11]. Studies of correlations between radon exhalation rate and temperature (at 1 m above ground) have given equivocal results.

61. The exhalation of thoron from soil is more dependent on soil conditions and meteorological factors than is the case for radon. Because of the short half-life of thoron, the effective exhalation depth is only a few centimetres and the thoron exhalation decreases rapidly when moisture content increases. Diurnal variations in thoron exhalation ($\pm 30\%$ of the mean) occur in dry summer days because of convection streams with maxima at sunset and at night and minima at sunrise and early in the day [D4, G16, M1, M10, S40].

62. Radon exhalation from sea water is about two orders of magnitude less than from soil. The radium concentration in sea water is about 1 Bq m^{-3} and varies with the depth, being higher (up to one order of magnitude) near the ocean floor than at the surface [S26]. The radon concentration, however, is not in equilibrium with radium in the sea water, but it is higher near the bottom owing to radon exhalation from the sea floor and to radium-enriched plankton in the water. This excess of radon decreases with increasing distance from the sea floor; in surface water there is actually a relative radon deficiency due to radon exhalation into the air. Measured values of radon concentration are about 1 Bq m^{-3} or less [H7, S26]. Using a diffusion coefficient of $10^{-5} \text{ cm}^2 \text{ s}^{-1}$ for radon in water leads to a predicted exhalation rate of about $10^{-9} \text{ Bq cm}^{-2} \text{ s}^{-1}$. However, owing to water turbulence, the effective diffusion depth is much higher than that derived from molecular kinetic diffusion and the radon exhalation is at least an order of magnitude higher than the above value. This higher value is also in better agreement with the ratio between the radon concentration in air above oceans and land, which is around 10^{-2} .

E. DISPERSION OF RADON AND THORON IN AIR

63. The transport and dispersion of radon and thoron in air depend on the vertical temperature gradient, the

direction and strength of the wind and on air turbulence. The radon and thoron daughters are also affected by precipitation. The temperature in the troposphere normally decreases with height up to about 11 km, above which the stratosphere begins. There the temperature is rather constant up to about 32 km, where it begins to rise again. Most of the air mass, water vapour and dust are found in the troposphere (75%), and for normal turbulence conditions most ($> 99\%$) of the radon and its daughter products are found in the troposphere.

64. The vertical dispersion of radon and thoron takes place through turbulent diffusion and convection and is limited by the radioactive decay. The turbulent diffusion coefficient is much higher than the gas kinetic diffusion coefficient and increases with the height above ground. This increase with altitude is influenced by vertical variations in wind velocity and atmospheric stability in the upper part of the troposphere and in the lower part of the stratosphere, where the turbulent diffusion coefficient decreases again [J3]. Characteristic profiles of the turbulent diffusion coefficient corresponding to some typical conditions of turbulence are shown in Figure X (adapted from [J3]).

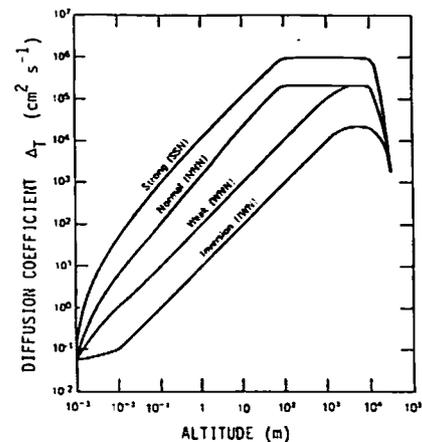


Figure X. Values of diffusion coefficient used in calculation of vertical concentration profiles of radon-222 and its daughters. NNN = normal turbulence conditions; WNN = weak turbulence conditions; SSN = strong turbulence [G19]

65. The vertical distribution of radon and thoron and their daughters in air can be calculated from the following system of differential equations

$$\frac{d}{dz} \left(\Delta_T(z) \frac{dX_{a,1}(z)}{dz} \right) - \lambda_1 X_{a,1}(z) = 0 \quad (17)$$

$$\frac{d}{dz} \left(\Delta_T(z) \frac{dX_{a,j}(z)}{dz} \right) + \lambda_{j-1} X_{a,j-1}(z) - (\lambda_j + \Lambda) X_{a,j}(z) = 0 \quad (j > 1) \quad (18)$$

where $\Delta_T(z)$ is the turbulent diffusion coefficient at the height z ; $X_{a,1}(z)$ is the concentration of radon or thoron atoms in air at the height z ; $X_{a,j}(z)$ is the concentration of daughter j ($j > 1$) in air at the height z ; λ_1 is the decay constant of radon or thoron; λ_j is the decay constant of radon or thoron daughter ($j > 1$); and Λ is the removal rate of daughters caused by washout and is assumed to be independent of altitude [J3]. Boundary conditions to equations (17) and (18) are $X_{a,j}(0) = 0$ for $j > 1$; and $X_{a,j}(\infty) = 0_{a,j}$ for $j = 1, 2, 3 \dots$

66. By assuming a constant radon and thoron exhalation rate from an infinite plane (ground surface) which equals the radioactive decay of the total radon and thoron content in the atmosphere, it is possible to solve equations (17) and (18), which, in combination with different values of $\Delta\tau(z)$, give the vertical distribution of radon and thoron and their daughters for different atmospheric stabilities. The result of such calculations is shown in Figure XI. It is seen that the radon and

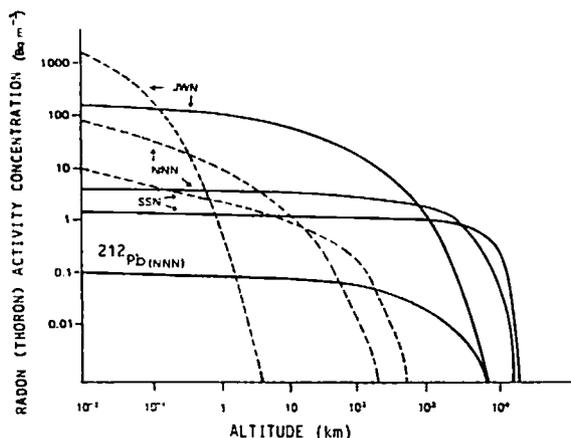


Figure XI. Vertical profiles of radon and thoron, assuming an emanation rate of $1 \text{ radon atom cm}^{-2} \text{ s}^{-1}$ corresponding to about $2 \cdot 10^{-2} \text{ Bq m}^{-2} \text{ s}^{-1}$ and $0.1 \text{ thoron atom cm}^{-2} \text{ s}^{-1}$ corresponding to about $1 \text{ Bq m}^{-2} \text{ s}^{-1}$. JWN = strong inversion; SSN = strong turbulence; NNN = normal turbulence. The full lines are for radon and the broken lines for thoron. The vertical profile of ^{212}Pb is also shown for normal turbulence conditions [J3]

thoron concentrations at ground level are expected to vary by a factor of 100 for the extreme conditions of atmospheric stability. Assuming a radon exhalation rate of $1 \text{ atom cm}^{-2} \text{ s}^{-1}$ (corresponding to $2 \cdot 10^{-2} \text{ Bq m}^{-2} \text{ s}^{-1}$), these variations of radon concentration lie between about 1 and 100 Bq m^{-3} . At higher altitudes these variations gradually reduce to become less than a factor of 5 at 1–2 km. Assuming a thoron exhalation rate of $0.01 \text{ atom cm}^{-2} \text{ s}^{-1}$ (which is a fairly normal value) the thoron concentrations at ground level (0.01–1 m) are about the same or a little higher than those of radon. At 10–100 m height the thoron concentrations are insignificant.

67. Several measurements have generally confirmed the above theoretical model for the vertical distribution of radon. Radon concentrations at different altitudes over the Yukon Valley in Alaska were found to lie within the theoretical curves for the extremes of strong inversion and normal turbulence conditions, as shown in Figure XII. The radon concentration in air below 2 km corresponds to a radon exhalation rate of $0.33 \text{ atom cm}^{-2} \text{ s}^{-1}$ (corresponding to $0.7 \cdot 10^{-2} \text{ Bq m}^{-2} \text{ s}^{-1}$), which is remarkably high as the ground was frozen and had a thick snow cover. This may be explained either by a high radium content in the soil or by a high diffusion coefficient for radon from the snow cover [L3]. The radon may also have come from elsewhere. Radon concentrations at altitudes above 2 km probably depend on large-scale circulation of air and radon transport from very distant areas, which gives time for substantial decay.

68. In measurements made by Bradley and Pearson [B8] over Illinois, United States, from 150 m to 5 km above ground the radon concentration ranged from 2 to 10 Bq m^{-3} at 150 m and decreased approximately by

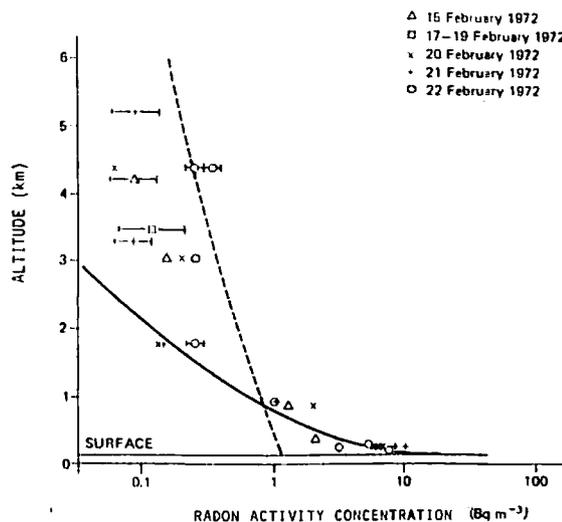


Figure XII. Radon concentration at different altitudes over the Yukon Valley in Alaska. The figure represents the average of 12 measurements over the Gulf and Alaska. The solid curve represents the theoretical profile of the exhalation rate of $0.33 \text{ atom cm}^{-2} \text{ s}^{-1}$ and a strong inversion. The broken curve represents theoretical profiles with normal conditions of turbulence [L3]

one-half for every 700 m increase in altitude. This vertical decrease with height is greater than theoretically expected from the air mixing conditions.

69. Even if some measured radon levels do agree with the models described above, the models should only serve as a guide to the understanding of radon levels. They are not adequate to explain all the variations in radon levels with geographical place, altitude and meteorological conditions. In practice, the radon exhalation rate varies with geographical location and the land surface is not unlimited, as the theory assumes.

70. Studies of the correlation between radon concentrations in air and meteorological conditions in different environments in France have been reported [F8, F9, G20]. Measurements of radon concentration at ground level during several months in combination with acoustic sounding measurements give a reliable method for the monitoring of vertical stability above a site [G21]. Radon concentration has also been used as an atmospheric tracer to follow air mass movements in urban pollution studies [C19].

71. Because of the short half-life of thoron, there are substantial difficulties in measuring this gas in the presence of radon in air. Therefore, actual data on thoron distribution with height over a few hundred metres to compare with the models are not available. Measurements on ^{212}Pb in air are sometimes incorrectly reported as thoron concentrations. As shown in Figure XI, equilibrium does not occur at ground level under normal turbulence conditions (NNN). Only in case of strong inversions (not shown in the figure) does equilibrium occur at ground level (1 m above ground) [J3]. There are however some measurements [D4, I5] of thoron (Figures XIII, XIVa) showing the vertical profile up to a few metres. Figure XIVb shows the corresponding profile for radon. The lower values observed during the day are caused by greater turbulence in daytime than at night [J5]. The results illustrated in these figures may depend on local factors and in general are not applicable to all sites and under all conditions.

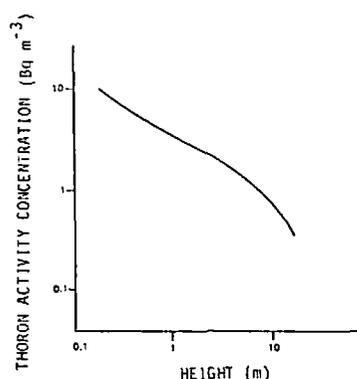


Figure XIII. Vertical profile of thoron in the atmosphere for wind speeds of 4 to 12 m s⁻¹ [D4]

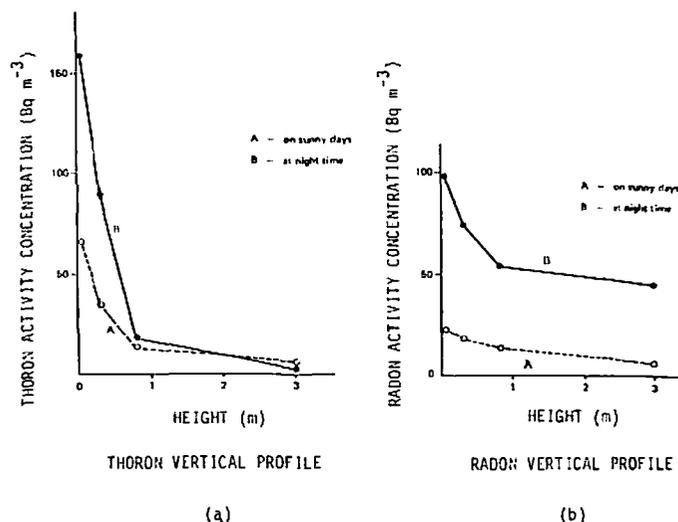


Figure XIV. Vertical profile of thoron (a) and radon (b) concentration near ground level [15]

72. A great discontinuity in the transfer of radon to air occurs at the boundaries of large land surfaces, e.g., a continent. Owing to the small radon emanation from sea water it is to be expected that the radon concentration in surface air near the coast should be much smaller when the wind blows from the sea than when it blows from the land. This effect is evident during sea breeze conditions in the day time when low radon levels occur, while off-shore wind conditions during the night usually bring higher radon levels [M15, S12].

73. Under conditions of constant wind velocity from the sea the growth of radon concentration near ground level at increasing distances from the coast is determined by the exhalation rate, local turbulent mixing and the radioactive decay but in practice it cannot be predicted by current models. Even at distances of several kilometres from the coast variations caused by wind direction occur, indicating a degree of non-equilibrium in the growth of radon concentration [M15].

74. With constant offshore wind velocity the radon and thoron concentration in air above the sea still decreases at increasing distances from the coast. This decrease is mainly caused by radioactive decay. Because of its short half-life the concentration of thoron decreases rapidly to insignificant values over the sea. On the other hand, continental air can still be identified several thousands of kilometres from the coast by its radon content [P11].

75. The concentration of radon in air over the ocean at large distances from land depends on the direction of the prevailing wind. A low radon concentration in "marine" air (some tens of mBq m⁻³), may rapidly increase by an order of magnitude or more in case of a change to "continental" air. Such "radonic storms" have been observed repeatedly [W8, L1]. Radon has therefore been used successfully as an indicator of the continental contribution to over-ocean air in studies on air pollution [L5, L4, W9, W10], and air transport mechanisms [W12].

76. Radon measurements during shipboard cruises show very low radon concentrations in maritime air (from less than a few mBq m⁻³ to about 0.1 Bq m⁻³) far from land. In smaller seas like the Mediterranean the

radon concentration in air may be of the order of 1 Bq m⁻³, indicating that this air contains much more continental radon than that over the open oceans [L10]. Along the coasts of large continents at 10–100 km distance, the radon concentration may still be of the order of 1 Bq m⁻³. The variations of the radon concentrations in air have also been demonstrated in the Arabian Sea and the Bay of Bengal [R10, E3]. The level of radon concentrations depended not only on the distance from land but also on changes in monsoon activity, causing varying degrees of interaction of continental and maritime air. Levels of ²¹²Pb are generally very low (less than 0.1 mBq m⁻³) [R10].

77. The radon concentration over an island depends on the radon exhalation rate from the ground and on the meteorological conditions. With no wind the radon levels are caused only by radon exhalation from the island itself. In windy weather the radon levels may be increasing inland, in the wind direction, and partly consist of radon from the island itself and partly of radon from a distant continent. This has been observed in measurements on Hawaii, where about 25% of radon on the lower slopes of the island was attributable to distant continental sources [M15].

78. The time-variation of radon and thoron concentration in the surface layer of the atmosphere depends on the variation of the radon and thoron exhalation rates and the vertical and horizontal dispersal of radon and thoron. Maximum exhalation is usually observed during the summer and minimum in the winter. However, vertical turbulent mixing is also higher in spring and summer, resulting in a decrease of the radon and thoron concentration at the surface. In autumn and winter this vertical exchange is smaller and inversions are frequent. The effect of decreased mixing may greatly outweigh the decreased exhalation rate, resulting in increased radon and thoron concentrations in autumn and winter, and the overall effect may be a seasonal variation with a minimum radon and thoron concentration in the spring and summer and a maximum in the autumn and winter. This has been observed by several investigators [M1, B6, R4, M5]. However, recent studies performed at Chester, New Jersey, United States, of the variation of radon concentration in air show a seasonal maximum in summer and a minimum in winter. This seasonal variation has been consistent throughout three years. Since the variations

are out of phase with the normal pattern of the distribution of inversion strength and there is a poor correlation between the radon exhalation rate and radon

concentration in air (at 1 m) no obvious explanation has yet been found for these seasonal variations [F5]. Figure XV shows the results, including studies on the

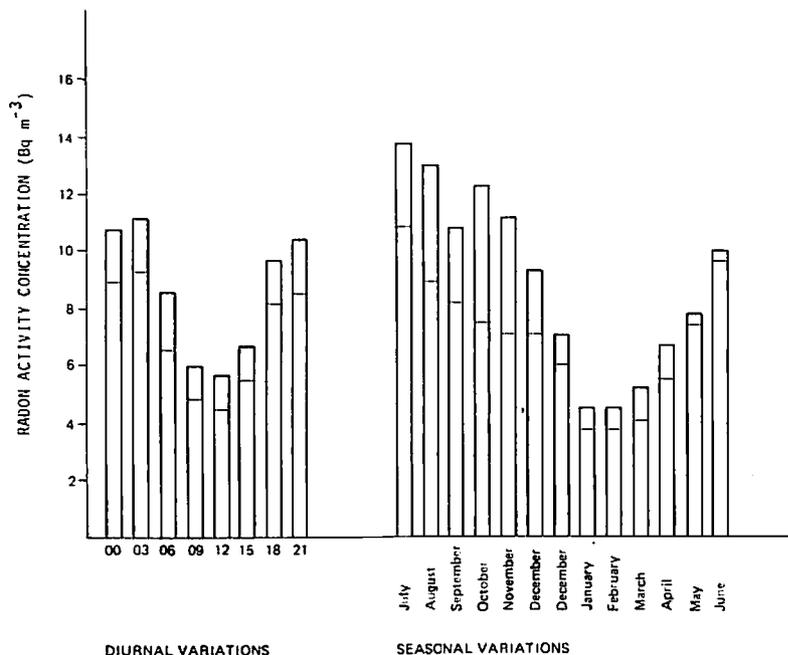


Figure XV. Variations in radon concentration. The diurnal variations show the annual means for the eight three-hour periods. The seasonal variations show the overall means for the thirteen four-week periods. The top lines in each column show the variations of the average for each observation period (1977/78, 1978/79, and 1979/80) [F5]

diurnal variations. The hourly data and three-hour averages were found to be log-normally distributed. The arithmetic mean and the median values (within parentheses) were 8.1 (6.3) Bq m⁻³ for 1977-1978, 8.5 (6.7) Bq m⁻³ for 1978-1979, and 7.0 (6.1) Bq m⁻³ for

1979-1980. Few thoron measurements exist to demonstrate its variations, but measurements of ²¹²Pb may indicate a similar behaviour by thoron. Examples of the variations of ²¹²Pb concentrations in air are presented in Figure XVI [B6].

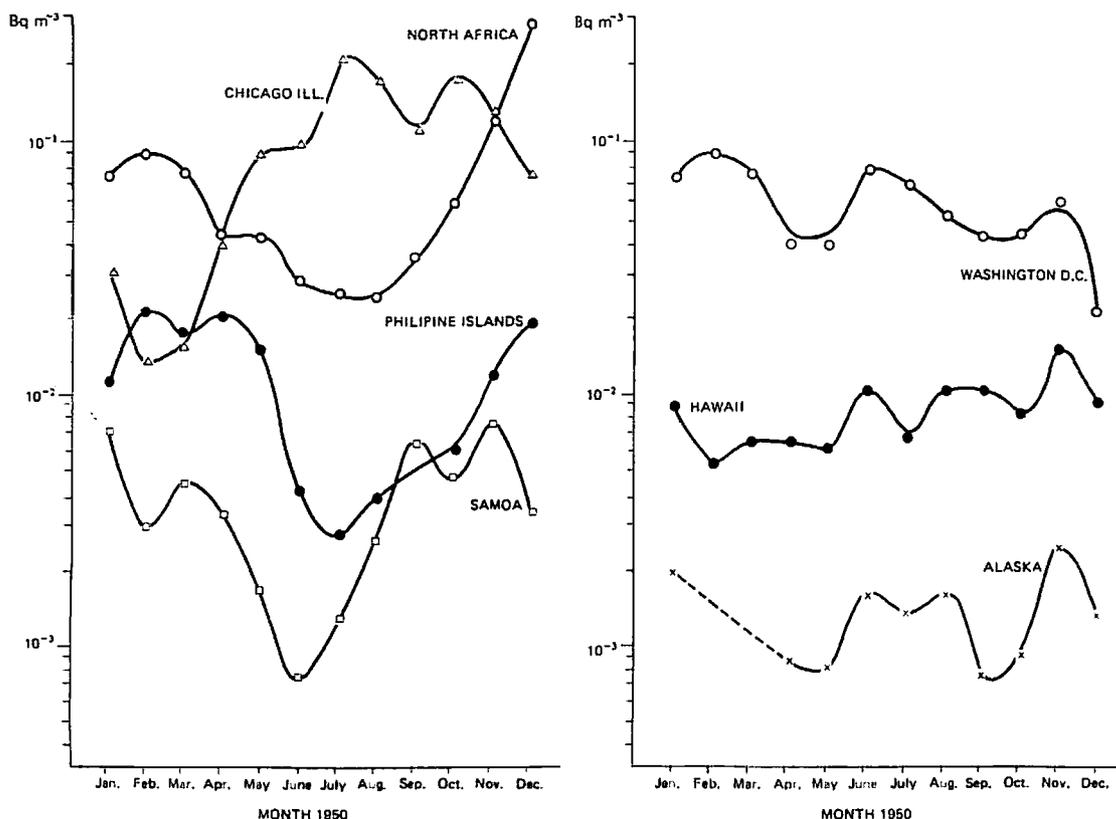


Figure XVI. Variations of the ²¹²Pb activity concentration in air, 1950 [B6]

79. Other extensive measurements in the United States of radon and thoron daughters were made over a period of 4 years in Cincinnati, Ohio in the early 1960s [G12]. Daily and seasonal variations were studied. Minimum radon daughter concentrations occurred in March and maximum values between August-October. The maximum levels were not so extended in time as they were at Chester, but the general shape is in fairly good agreement.

80. Several measurements of diurnal and seasonal variations of radon concentration in air in France have been reported [F20, D13]. The diurnal variations show a minimum at noon and a maximum at midnight and a decrease with increasing altitude (0-100 m) [D13]. Seasonal variations of thoron and thoron daughters at

ground level have also been measured [F21] and show a maximum in March-April and a minimum in December-February. The variations were within a factor of ten and the concentrations of thoron lay in the range 2-20 Bq m⁻³ for thoron and 0.1-1 Bq m⁻³ for thoron daughters. Measurements of seasonal and diurnal variations of radon, radon daughters and ²¹²Pb have also been reported from Spain [G22, G23]. Maxima were observed in the winter and minima in the summer and the concentrations were 2-3 times lower in the afternoon than in the morning.

81. Another example of strong seasonal variations was found by Rangarajan et al. [R4]; it is illustrated in Figure XVII. Since these measurements were made on

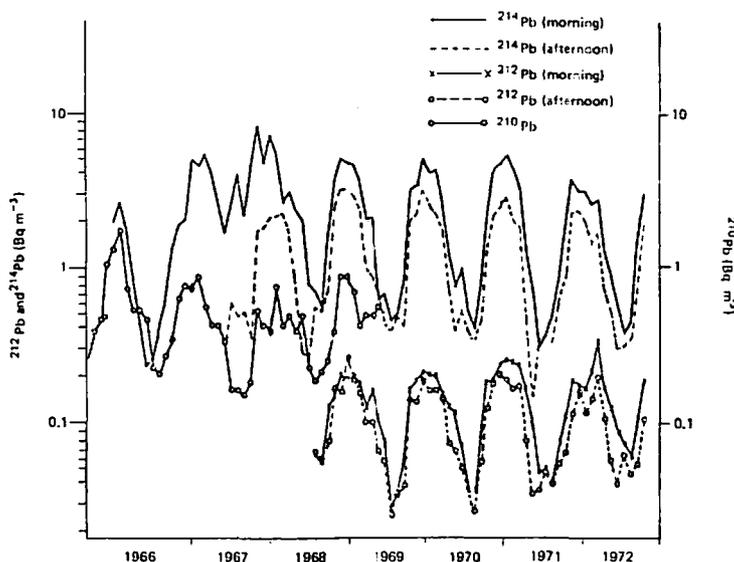


Figure XVII. ²¹⁴Pb, ²¹²Pb and ²¹⁰Pb activity concentrations in surface air in Bombay (monthly mean concentrations) [R4]

²¹⁴Pb and ²¹²Pb, the variations do not quite represent radon and thoron variations; nevertheless, they are of interest because they are a consequence of several contributing factors. The minimum in the spring and summer is caused by increased turbulent mixing, a predominant south-west wind over the Indian Ocean, higher mean wind speeds and intensive rainfall which affects both the ²¹⁴Pb concentration in air and the exhalation rates. Seasonal variations are less pronounced in areas with cold winters and an early snow cover, which greatly reduces the exhalation of the two gases. This has been demonstrated by the results of measurements made in Finland [M5] and at a high altitude station in India [R4]. Diurnal variations are mainly caused by variations in turbulent mixing.

82. Radon and thoron concentrations reach their maximum in the early morning and their minimum at noon or in the afternoon [R4, F5] and the variation is generally less than one order of magnitude. The results of repeated measurements are shown in Figure XV [F5]. Diurnal variations of radon concentration do not occur over the oceans [E3].

83. Dispersion of radon released from point sources such as geothermal power plants, mine ventilation outlets, etc., or from sources with a small area extension follows the normal dispersion kinetics of gases. If the source may be considered approximately as a point

source, the concentration in the environment may be estimated by using the ordinary dispersion formula for atmospheric diffusion.

84. The average concentrations per unit release rate (also called "dispersion coefficients") at various distances from a "point" source have been calculated for normal conditions in the United Kingdom as mean concentrations weighted for the frequency of different meteorological conditions [B12, C20]. The results (based on [C20], fig. 36) are shown in Figure XVIII, curves 1a, 1b and 1c. The release point is assumed to be 0, 30 and 100 m, respectively, above ground and the deposition velocity to be zero. Radon releases from extended sources such as tailings can be considered as a number of small point sources distributed over the area of interest. The dispersion of radon from such sources has been calculated for two area sources, one 1 x 1 km² (10⁶ m²) and another 100 x 100 m² (10⁴ m²). The surface was divided into 250 and 100 equal elements, respectively, and the resultant dispersion coefficient at different distances from the outer frontier of the surfaces was calculated using curve 1a in Figure XVIII for each surface element. The values of the horizontal dispersion parameter σ_y have been calculated assuming a Pasquill D weather category and the formula $\sigma_y = 0.1471 x^{0.9031}$, where x, in metres, is the distance from the source. The results are presented in Figure XVIII, curves 3 and 2, respectively.

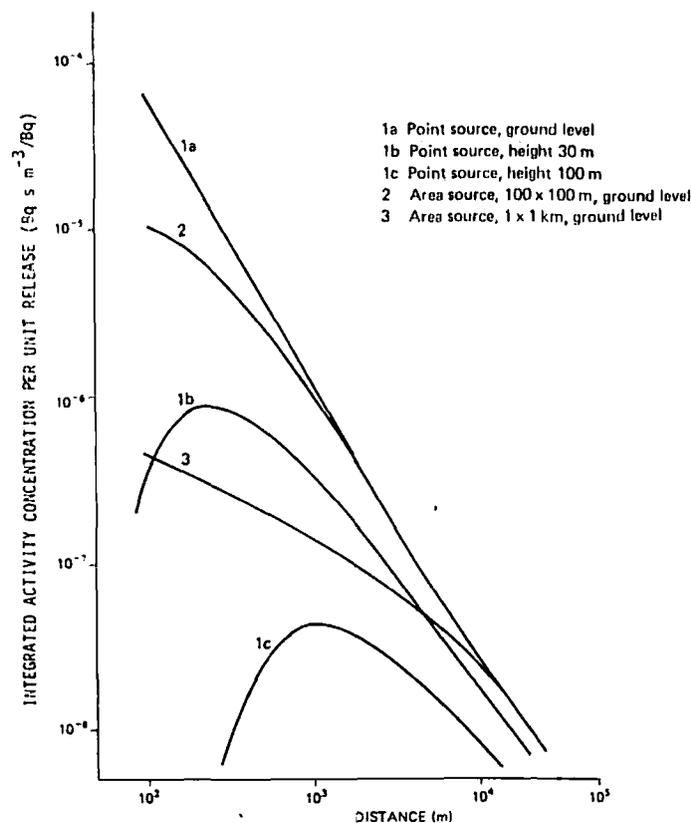


Figure XVIII. Integrated activity concentration of radon per unit release at various distances from point and area sources, calculated for normal conditions in the United Kingdom. The values are given as mean concentrations weighted for the frequency of different meteorological data as given in [C20]

85. The average concentrations per unit release rate taken from Figure XVIII curves 1a and 1b, for point sources are approximately inversely proportional to the distance x raised to the power p . If the activity concentration $\chi_a(x)$ at distance x is expressed relative to the concentration $\chi_a(x_0)$ at a reference distance x_0 , $\chi_a(x)$ is given by

$$\chi_a(x) = \left(\frac{x}{x_0}\right)^{-p} \chi_a(x_0) \quad (19)$$

With $p = 1.5$ the formula describes satisfactorily the dispersion and relative concentration at distances beyond 1 km from a point source (curve 1a, Figure XVIII) at ground level, with the reference distance $x_0 = 1$ km. For releases from a height of 30 m or more a value of $p = 1.2$ approximates more closely the dispersion curves at distances beyond 1 km.

F. DISPERSION OF RADON AND THORON DAUGHTERS

86. The atmospheric diffusion model of Jacobi and André has also been applied to radon and thoron daughters [J3]. In the case of radon, the concentration at a given point and its vertical distribution are greatly dependent on the conditions of turbulence in the troposphere. At 1 m from the ground the relationship between the activity of radon daughters and radon depends on the turbulence but above 100 m the model predicts approximate radioactive equilibrium. This is also the case for ^{212}Bi relative to ^{212}Pb . Irrespective of turbulence conditions there is no general equilibrium between thoron and its daughters. Other similar models

by Staley [S20] and Birot et al. [B5] include time variations for diffusion and dispersion by advection. Shapiro et al. have developed these models with some modifications of the boundary conditions [S7].

87. Measurements in the lower atmosphere indicate deviations from secular equilibrium for radon daughters. Shapiro et al. [S7] have measured the $^{214}\text{Bi}/^{214}\text{Pb}$ ratio at 20 m under various atmospheric conditions. They have found that afternoon samples taken for a period of several months had $^{214}\text{Bi}/^{214}\text{Pb}$ activity ratios from 0 to 2.2 with a mean value of 0.67 (morning samples had a mean value of 0.70). The predicted range of the $^{214}\text{Bi}/^{214}\text{Pb}$ activity ratios according to Jacobi and André [J3] is 0.68–0.98 at a height of 20 m. "Abnormal" ratios greater than 1, which were all associated with winds greater than 4 m s^{-1} are outside the framework of any simple theory since different mechanisms must be responsible for these abnormal conditions. Possible explanations could be selective removal of radon daughters (e.g., by washout) and different responses to advection owing to attachment to aerosols of different size ranges [S8]. Other measured values of the relative concentrations of radon daughters range from 0.8 to 1 for heights between 1 and 90 m (increasing with height) [H15, H16, G11].

88. Changes in the radon daughter concentration profiles may influence the natural gamma-ray background. Thus, after heavy rainfall there is a depletion of radon daughters in air and a consequent increase of radon daughters on the ground. The increase in external absorbed dose rate in air ranges from $0.5 \cdot 10^{-2}$ to $4 \cdot 10^{-2} \mu\text{Gy h}^{-1}$ [B3, F2], to be compared with a "normal" background of $5 \cdot 10^{-2} \mu\text{Gy h}^{-1}$ (Annex B).

89. The occurrence and behaviour of the relatively short-lived radon daughters (^{218}Po , ^{214}Pb , ^{214}Bi and ^{214}Po) in the atmosphere are strongly dependent on the build-up from radon and by their short half-lives. Except for the circumstance of rainout, the radioactive decay time of these nuclides is short compared with any other removal mechanisms.

III. RADON AND THORON IN INDOOR SPACE

90. There is in principle no basic difference in the formal treatment of radon and thoron in a room or in the open air. However, there are other sources and factors which warrant separate discussion. The indoor space may be a house, apartment, mine, cave, tunnel or any other closed space, with or without ventilation. As for outdoor air, the levels of thoron and thoron daughters are low in comparison with those of radon and radon daughters. There are also very few data reported in the literature for thoron and its daughters and for this reason the emphasis of this chapter will be on radon and its daughters.

A. SOURCES

91. Radon and thoron in indoor space originate from emanation of the gases by the walls, floor and ceilings which are constructed of building material, rock or soil, by release from materials brought into the room, such as radon-rich water or gas and by radon or thoron in the inlet air, which may in turn have a normal concentration of the gases or an increased concentration derived from sources outside the room. The primary sources are in all cases ^{226}Ra or ^{224}Ra .

92. The radium and thorium concentration in building materials vary for different kinds of material, depending on their origin. Some published data are found in Table 10. More detailed information for building materials in Nordic countries is given in Table 11. The values in the tables are mean activity concentrations. Some materials such as phosphogypsum, aerated concrete with alum shale and uranium mine tailings, are known to be much more radioactive than others: some of them are used only in exceptional circumstances. Materials with low activity concentrations include wood and natural gypsum. The average values of radium and thorium concentrations in the above tables are of the order of 50 Bq kg^{-1} . In general the activity distribution of samples of individual building materials reflects the activity distribution of constituents of the building materials from different parts of a country. In an investigation of radium content of gravel in 23 of 24 provinces in Sweden, the standard deviation of all 146 samples was 30% and the range was within 0.5 and 2 times the average. The standard deviation for samples from each province was typically 20% [H1].

93. The highly radioactive materials are with few exceptions by-products from other processes. Phosphogypsum is produced in the manufacture of phosphoric acid from sedimentary phosphate ore; red mud bricks contain a waste product from the production of alumina from bauxite; blast-furnace slag is a by-product of iron production; fly-ash is a waste product from the combustion of coal; tailings are wastes from uranium mining and milling. In recent years there has

been increasing interest in using waste products as substitutes for natural products in building materials. However, because of the relatively high radioactive contents their use is frequently subject to regulation [W20].

94. 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 market for building materials in Sweden. Production was stopped completely 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.

95. The soil beneath a house can be a significant source of radon for individuals [O5]. This may be specially true if there are cracks in the base structure. The radium concentration in soil and rocks varies widely in different countries and in different regions of a country. Special areas having enhanced or naturally high radium concentration are some reclaimed lands in the United States following phosphate rock strip mining [U2], tailings from a radium factory in Australia [A8], uranium mine tailings in Colorado, United States [C16], and areas generally adjacent to uranium mining. The contribution from radium in the soil to the radon concentration in a house depends on the emanating power and the thickness and tightness of the base structure. In residential buildings in New Jersey and New York, United States, the radon exhalation rate per unit area from cellar floors has been determined. The average exhalation rate per unit of radium activity concentration ($\text{Bq m}^{-2} \text{ s}^{-1}$ per Bq Ra g^{-1} soil) for 16 houses was 0.07 ± 0.06 with a range of 0.007–0.21 [G1]. The thickness of the concrete floor required by local building codes is about 10 cm. The wide variations observed may have been due to differences in emanating power of the underlying soil. There may also have been a contribution from the concrete floor itself and from deeper layers of the soil. Bare soil in unpaved crawl spaces has been identified as a major radon source in some houses in the United States [R11]. In Sweden the soil under the house has recently been proved to be a more significant source of radon than building materials, both in cases of very high individual radon concentrations and for general radon exposure of the public [R15]. This is also confirmed by the general observation that the radon concentrations in high floors of multi-storey buildings tend to be lower than in ground floors.

96. Another source of indoor radon is radon-rich water. The release of radon from water to air depends on the circumstances in which it is used and the resulting radon concentration in air is determined by the ventilation rate. The radon release during typical household activities has been studied [P2] and for a house with volume 340 m^3 and ventilation rate of 1 h^{-1} the average air-to-water concentration ratio has been estimated to be $0.4 \cdot 10^{-4}$. The amount of water used each day was assumed to be about 500 l per person. The air-to-water concentration ratio in dwellings in Finland has been measured to be about 10^{-4} [C3]. In other studies the ratio has been calculated to be $1.5 \cdot 10^{-4}$ if full desorption of radon in water is assumed [N2] and 10^{-4} by another research team [D14]. In Annex B of the UNSCEAR 1977 report [U6] the ratio was calculated to be $2 \cdot 10^{-4}$. The ratio has also been expressed as a formula [G5]

$$\frac{\chi_{a, Rn}}{\chi_{w, Rn}} = \frac{\sum_k F_{a, Rn, k} V_{w, k}}{24 \lambda_w V} \quad (20)$$

where $\chi_{a,Rn}$ and $\chi_{w,Rn}$ are the radon concentrations in air and water, respectively; $V_{w,k}$ is the volume of water used daily for various applications k ; $F_{a,Rn,k}$ is the fraction of radon released to air for each application k ; λ_v is the ventilation rate of the dwelling h^{-1} ; and V is the volume of the dwelling. If $\sum_k F_{a,Rn,k} V_{w,k} = 1 \text{ m}^3/d$, $\lambda_v = 1 \text{ h}^{-1}$ and $V = 200 \text{ m}^3$, the ratio is about $2 \cdot 10^{-4}$. Using this ratio and taking typical values of the radon concentration in air ($10\text{--}100 \text{ Bq m}^{-3}$), it is apparent that radon in water is a significant source for radon in houses only when the radon concentration in water is of the order of 10 kBq m^{-3} or more.

97. Natural gas containing radon may also be a source of significance. The industrial processing of raw natural gas involves the removal of impurities and separation of hydrocarbons. Some of these hydrocarbons (mainly propane) are bottled under pressure for sale as liquefied petroleum gas (LPG), while the others are used for fuel or as chemical feed stocks. When either gas is burned in houses, radon is released and enhances the radon level indoors [G6]. The contribution from this source depends on the amount of gas used, the original radon concentration and the elapsed time between production and consumption.

98. Radon enters the natural gas in the earth by diffusion from radium deposits. Uranium minerals are often associated with carbonaceous deposits and radon in natural gas is therefore not unexpected. Radon diffuses through the rock along with the natural gas to collection wells, from which the gas (and radon) is transported to the gas processing plant. The radon concentration in raw natural gas at production wells varies from undetectable values up to levels of the order of 50 kBq m^{-3} [U6, H8] (see Table 12).

99. The processing of raw natural gas results in the transfer of 30–75% of the radon from the natural gas into the LPG. The remaining radon may be further diluted if the gas becomes mixed in transit with gas from other wells with low radon concentration. During transport through long transmission lines and storage in reservoirs the radon concentration in natural gas in the distribution system is reported to lie within the range of about $0.04\text{--}2 \text{ kBq m}^{-3}$ [W17, J8]. The average value for the United States is estimated to be 0.7 kBq m^{-3} , except for four States which have an average of about 2 kBq m^{-3} [B1].

100. The gas is used for kitchen appliances and domestic heating. Taking representative consumption levels of $0.8 \text{ m}^3 \text{ d}^{-1}$ and $2.8 \text{ m}^3 \text{ d}^{-1}$, respectively, and a mean radon concentration of 0.7 kBq m^{-3} , the mean activity introduced in homes is 2500 Bq d^{-1} (with a range of $130\text{--}6500 \text{ Bq d}^{-1}$ [J8]) if the appliances are unvented. If the combustion products are vented outside the house, this radon source is negligible.

101. The radon concentration in LPG depends on the radon concentration at the well, the transit time from the well to the processing plant, the composition of the inlet gas and the type of processing. The overall effect is an increase in the concentration of radon in the end product LPG and an enrichment factor of 10 ± 5 with respect to raw natural gas has been reported [G6].

102. However, the radon concentration in LPG delivered to the consumer depends, in addition, on the time delay in a complex network of supply, transit, storage and delivery facilities which vary in different countries and different parts of a country. Because of

decay of radon the concentration may be lower in remote districts than in districts close to the processing areas. Seasonal variations may also occur owing to different storage times, with a minimum radon concentration in the winter when gas consumption is high. Seasonal variations in concentration of about a factor of 2 have been found [G6].

103. During storage of LPG in domestic tanks the radon concentration is further reduced by decay. With monthly deliveries the average concentration of radon at the point of gas consumption is about 20% of the concentration at delivery.

104. The magnitude of this radon source in homes is estimated by combining the radon concentration in LPG delivered to homes, the quantity of LPG used and the decay factor due to storage in domestic tanks. In one investigation in the United States the values of these quantities have been found to be the following [G6]: radon concentration = $0.4\text{--}5.6 \text{ kBq m}^{-3}$; quantity of LPG = $0.3 \text{ m}^3 \text{ d}^{-1}$ for cooking ranges and $1.1 \text{ m}^3 \text{ d}^{-1}$ for space heaters; decay factor = 0.18. For normal consumption of LPG, the range of radon activity introduced into homes is then approximately $100\text{--}1500 \text{ Bq d}^{-1}$ with a customer-weighted average of about 200 Bq d^{-1} . The use of LPG is however small compared with natural gas and, on a national basis, natural gas is a 10–100 times greater radon source than LPG.

105. The relative significance of different radon sources for a reference house, defined as having a volume of 200 m^3 and an inner surface area of 350 m^2 , are presented in Table 13 (the values refer to data in this report). For thoron and thoron daughters, the only significant sources are building materials and the outside air. The thoron exhalation from building materials may be up to two orders of magnitude higher than that of radon, but in practice wall paper and other sealants may reduce the thoron exhalation considerably.

106. In mines, tunnels, caves and other underground buildings the radon sources are rocks, soil and water. Faces of uranium ore in mines are apparently potential radon sources, although there are great variations in emanating power. Crushed rock in abandoned parts of a mine is also a significant radon source in non-uranium mines [S12]. Radon in water may often be a significant source. Drainage or use of large amounts of radon-rich ground water may cause radon release either locally or to the ventilation inlet air. This may be the case in mines, tunnels, caves, spas and underground hydroelectric power stations [S13]. With poor ventilation local radon and radon daughter concentration in air may become very high if the radon concentration in water is $0.1\text{--}1 \text{ MBq m}^{-3}$ or more [S12].

B. RADON AND THORON DIFFUSION AND EXHALATION

107. The diffusion through and exhalation of radon and thoron from building materials, rock walls, etc., follow the principles described in paragraphs 56–62. The emanating power F_r varies for different materials. Normally the value of F_r varies between 1% and 10% although extreme values of 0.1% and 30% have been reported [P4, W4].

108. The diffusion coefficient in a concrete building material depends on the type of aggregates used in the

concrete, the water/cement ratio used in the mix and the curing conditions. The diffusion coefficient can be expressed as Δ_k or as $\Delta_{eff} = F_{build,ps} \cdot \Delta_k$, where $F_{build,ps}$ is the porosity. Δ_k is called the effective diffusion coefficient and Δ_{eff} the effective bulk diffusion coefficient. The presentation of diffusion coefficients in the literature is not always very clear on this point and misunderstandings arise easily. This problem is discussed by Culot et al. [C12]. Another name used in the literature for effective bulk diffusion coefficient is true diffusion coefficient.

109. Another source of error arises from the experimental conditions under which the diffusion coefficient is measured. By placing the material of interest in a closed vessel and measuring the equilibrium radon activity in the air, the estimated diffusion coefficient is lower than that corresponding to emanation into open air [J9]. This is due to the fact that the difference in partial pressure of radon in the pores of the material and in the air space of the vessel decreases with time and this influences the rate of diffusion.

110. The exhalation of radon and thoron from walls is dependent on the ^{226}Ra and ^{224}Ra concentration in the walls, the emanating power and the diffusion coefficient. The diffusion coefficient for building material varies by several orders of magnitude for different materials and values between 10^{-4} and $10^{-7} \text{ cm}^2 \text{ s}^{-1}$ have been reported [C12, J9, P8]. Changes of air pressure have been shown experimentally to influence the radon exhalation rate from concrete walls [M8]. On reducing the atmospheric pressure, the exhalation rate was found to increase linearly with the pressure drop, as seen in Figure XIX. This effect has also been found in uranium

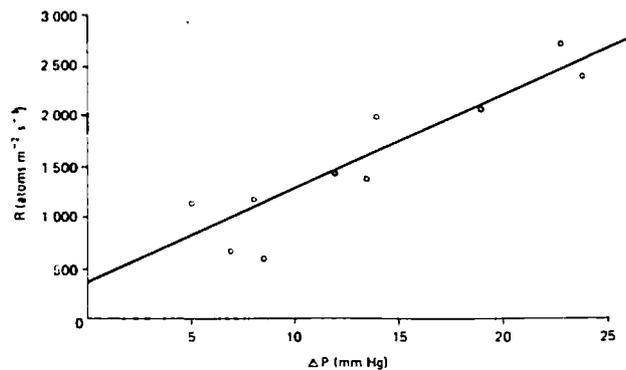


Figure XIX. Variation of the exhalation rate R as a function of the reduction of air pressure ΔP [M8]

mines and has been explained as the result of increased air flush through radon-rich cracks and fissures into the mine [P5, P6].

111. The exhalation rate of radon from building materials has been studied extensively. However, the values obtained are not always consistent and large variations are found for the same material and surface. There are obviously great experimental difficulties and factors influencing the measurement are not fully understood. Reported values of the radon exhalation rate per unit concentration of ^{226}Ra in building materials normally lie in the range of 10^{-6} to $10^{-4} \text{ Bq m}^{-2} \text{ s}^{-1}/(\text{Bq kg}^{-1})$ with several values in the higher part of the range [G1, P8, J9, S25, M17, W4].

112. Radon emanation can be reduced by applying a sealant to the emanating surface. This is of particular interest for houses built on uranium tailings or other

waste products. Although promising results have been achieved, there is not yet full acceptance of the methods proposed, partly for economic reasons and partly for uncertainties regarding the overall positive effect. A sealant may in itself cause harm by being toxic or electrostatic, etc. A possible negative effect is the increase in gamma radiation from the daughters trapped behind the sealant. This depends on the material and its thickness and may be as high as 10%. Data have been published on the qualities of some radon barriers [C14, H3] and on the experience gained in practical attempts to reduce radon and radon daughter concentrations indoors [C15]. By plastering the wall the radon exhalation may even increase if the radium concentration in the plaster is higher than that in the wall [W4]. By covering the walls with plastic materials such as polyamide, polyvinylchloride, polyethylene and epoxy paint (the thicknesses of which are of the order of 0.1 mm) or by painting three times with oil-based paints, the radon exhalation is reduced by one order of magnitude [E4, P13]. The effect of wall paper is always to decrease radon exhalation (e.g., 30% as given by [O3]), even if it is sometimes marginal. The reduction is probably higher for thoron, because of its short half-life.

113. The emanation of thoron from walls depends on ^{224}Ra concentration in the wall, the emanating power and the diffusion coefficients. Owing to the short half-life of thoron, the emanation may be considered as taking place only from the outer surface layers. There are very few measurements of the exhalation rate of thoron, because thoron daughters in houses are believed to be less important than radon daughters as a potential health hazard. There are also experimental difficulties in making such measurements. Experimental data have been reported [M9] on thoron exhalation rates from building materials, relative to exhalation rates of radon. Values of 0.1–0.4 ($\frac{\text{Bq of thoron}}{\text{Bq of radon}}$) were observed.

C. RADON AND THORON DISPERSION IN INDOOR SPACE

1. Indoor spaces other than tunnels

114. Assuming instantaneous mixing of radon or thoron in a room, there will be a homogeneous activity distribution and the concentration in a room can be found by solving the following equation

$$\frac{d\chi_a(t)}{dt} = R \frac{S}{V} + \frac{A_k}{V} + \chi_{a,in} \lambda_v - \chi_a(t) (\lambda + \lambda_v) \quad (21)$$

where $\chi_a(t)$ is the radon or thoron activity concentration in the room at the time t ; R is the radon or thoron activity exhalation rate per unit area; S is the emanating surface area; V is the volume of the room; A_k is the activity release rate of any other source in the room (water, gas); $\chi_{a,in}$ is the radon or thoron activity concentration in the inlet air; λ_v is the ventilation rate (h^{-1}); and λ is the decay constant of radon or thoron.

115. At equilibrium the concentration in the room is given by

$$\chi_a = \frac{R \frac{S}{V} + \frac{A_k}{V} + \chi_{a,in} \lambda_v}{\lambda + \lambda_v} \quad (22)$$

In homes $0.1 < \lambda_v < 3 \text{ h}^{-1}$. Since for radon $\lambda = 7.6 \cdot 10^{-3} \text{ h}^{-1}$, $\lambda_v \gg \lambda$ and expression (22) takes the form

$$\chi_{a, \text{Rn}} = \frac{R \frac{S}{V} + \frac{A_k}{V}}{\lambda_v} + \chi_{a, \text{Rn}, \text{in}} \quad (23)$$

As long as $\lambda_v \gg \lambda$ and $\chi_{a, \text{Rn}, \text{in}}$ is negligible, the radon concentration indoors is inversely proportional to the ventilation rate. As the ventilation rate increases from 0 to 0.1 and from 0.1 to 1 h^{-1} , the radon concentration decreases by a factor of 13 and 10, respectively. For thoron $\lambda = 45 \text{ h}^{-1}$ so that $\lambda \gg \lambda_v$ and expression (22) takes the form (A_k is assumed to be negligible)

$$\chi_{a, \text{Tn}} = \frac{R \frac{S}{V} + \chi_{a, \text{Tn}, \text{in}} \lambda_v}{\lambda} \quad (24)$$

This means that the thoron concentration in room air is almost independent of the ventilation rate, as long as the contribution from outdoor air is small. The concentration of ^{212}Pb , on the contrary, is inversely proportional to the ventilation rate, down to about 0.5 h^{-1} and thereafter it approaches the activity concentration of thoron.

116. The numerical value of $\chi_{a, \text{Rn}}$ for radon in a reference house such as that described in paragraph 105 is

$$\begin{aligned} \chi_{a, \text{Rn}} &= \left(2 \cdot 10^{-3} \text{ Bq m}^{-2} \text{ s}^{-1} \cdot 3600 \text{ s h}^{-1} \cdot 1.75 \text{ m}^{-1} + \right. \\ &+ \left. \frac{4 \cdot 10^3 \text{ Bq d}^{-1} + 3 \cdot 10^3 \text{ Bq d}^{-1}}{24 \text{ h d}^{-1} \cdot 200 \text{ m}^3} \right) / 0.5 \text{ h}^{-1} + \\ &+ 4 \text{ Bq m}^{-3} = \frac{12.6 + 1.45}{0.5} + 4 = 32 \text{ Bq m}^{-3} \end{aligned}$$

The relative importance of radon sources may be different in different parts of a house and in case of inhomogeneous mixing of air in the building the radon concentration will vary within the house. In the cellar the radon exhalation rate is often higher than in the upper floors. The ventilation rate is also often less and the radon concentration is therefore normally higher in the cellar than in the upper floors. Water as a radon source is of significance in the kitchen, bathroom and laundry. It contributes specifically to radon daughter exposure because it becomes available as a source when people are using it.

117. The variations of the radon concentration in a building are mainly dependent on the variations of the ventilation conditions and air exchange between rooms. These are caused by meteorological conditions (wind, barometric pressure, temperature) and by human activities such as opening windows and doors. Some of these effects are shown in Figures XX and XXI [S30]. There are also other variations caused by changes of the radon exhalation from surfaces. The variations can depend on pressure change (see paragraph 110) and a correlation between radon concentration in a room and barometric pressure has also been observed [J10].

118. Diurnal variations of the activity concentrations of radon, thoron and their decay products have also been measured [P14]. These variations follow the same pattern but the relative change is much smaller for

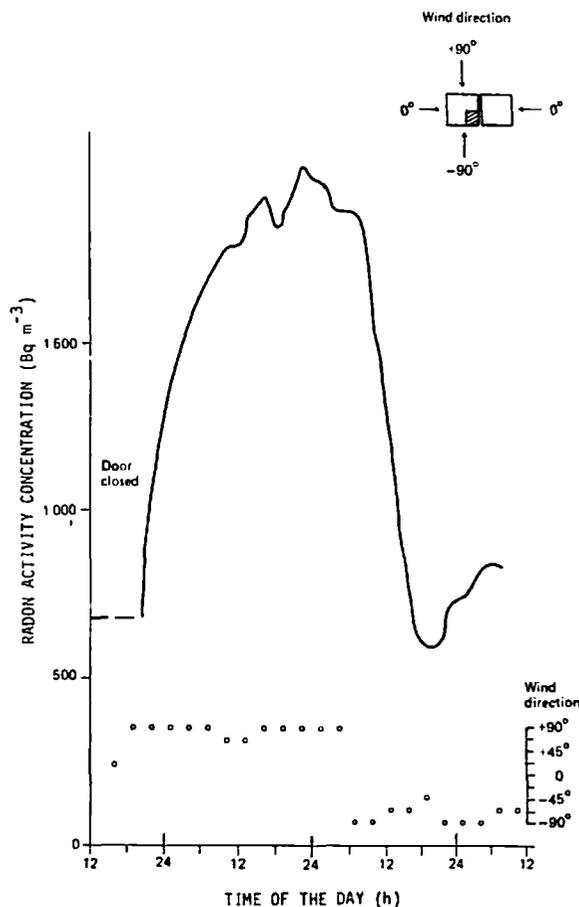


Figure XX. The dependence of the radon concentration in a detached house with a natural draught ventilation system on the wind direction. The door to the room was closed at the beginning of the measurement. (Measurements taken in winter) [S30]

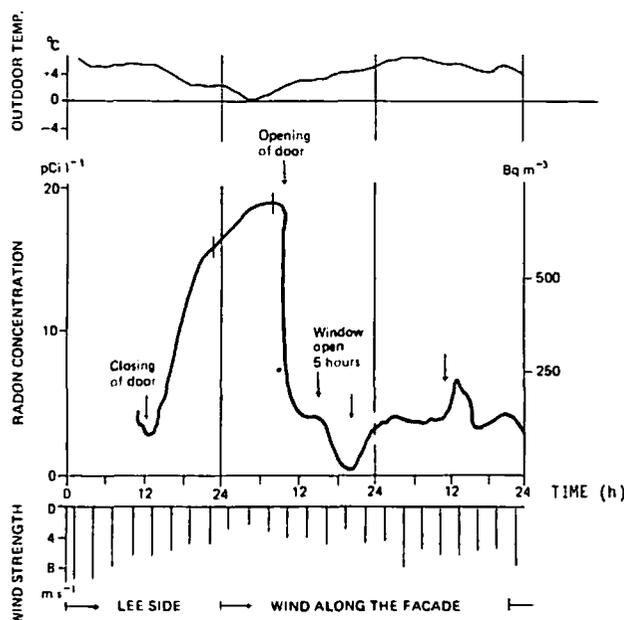


Figure XXI. Variation of the radon concentration in an apartment house built of elements of both ordinary concrete and aerated concrete based on alum shale and with forced ventilation system for the exhaust air. (Measurements taken in winter) [S30]

thoron than for radon, as expected from equation (24) and as can be seen in Figure XXII. This figure also illustrates the effect of changes in ventilation conditions.

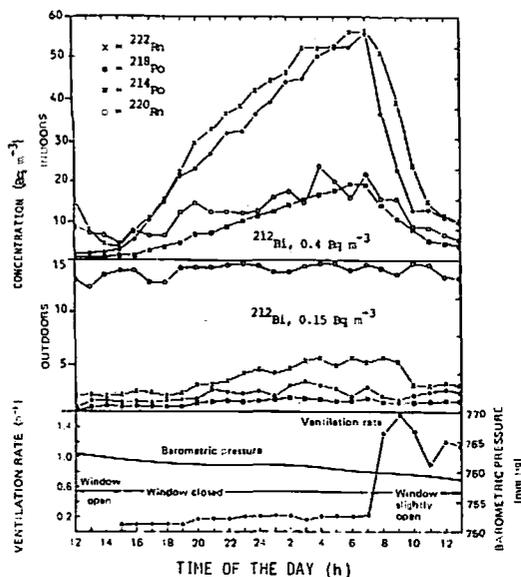


Figure XXII. Diurnal variation of concentration of radon, thoron, and their decay products indoors and outdoors of a typical dwelling [P14]

119. A study of the correlations between concentration of radon, radon daughters and thoron daughters and meteorological variables has been made in Innsbruck, Austria [S21]. About 750 measurements were

made in 12 rooms and the concentration data together with 24 meteorological variables were used for regression analysis.

120. Tables 14 and 15 show the results of activity measurements in the houses, taken from this study [S21]. $\Delta\chi$ is the largest observed increase of the activity concentration within 24 hours. The mean radon concentration in the open air of Innsbruck was 13.3 Bq m^{-3} with a maximum of 96 Bq m^{-3} . The mean radon activity concentration in living and working rooms ranged from 28 Bq m^{-3} to 115 Bq m^{-3} with a maximum value of 276 Bq m^{-3} . The mean thoron activity concentration in all rooms ranged from 3.7 Bq m^{-3} to 41 Bq m^{-3} . About the same range of thoron concentrations were found in the open air near ground level (1.5–1 m). The observed concentrations of ^{212}Pb were all considerably lower than those of thoron, with mean values between 0.3 Bq m^{-3} and 4.7 Bq m^{-3} . The mean activity concentration ratio $^{212}\text{Pb}/^{220}\text{Rn}$ was 0.019 (range 0.009–0.49). Table 16 summarizes the results of the regression analysis for radon and thoron with the statistically significant meteorological variables. Other variables, such as mean cloud cover, daily maximum wind speed, change of temperature gradient between morning and noon and noon to morning, and all other variables with time scales greater than 24 h were found not to be significant.

121. Measurements made in Norway [S24] (Figure XXIII) also show the variation of radon concentration

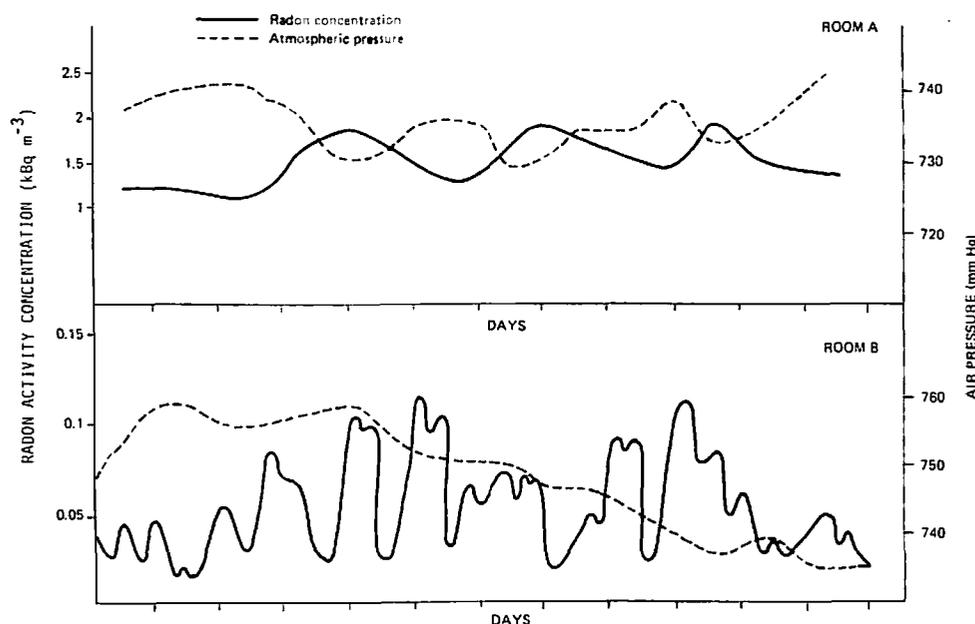


Figure XXIII. The variation of radon concentration in an unventilated room (A) and in an artificially ventilated room (B). Measurements taken in winter (A) and spring (B) [S24]

in rooms as a function of the atmospheric pressure changes. Room A had poor ventilation and room B had artificial ventilation which was different for day and night. This explains the great diurnal variations.

122. In houses built with a crawl space or with radium-rich waste products as filling material underneath the house, the radon concentration can change substantially with meteorological conditions [S31]. This is believed to depend on the variations in the pressure

difference between the house and the crawl space and on the variation in the ventilation of the crawl space itself. In a special study of the influence of wind strength and direction [W2] it was found that in ordinary houses radon may decrease normally by a factor of 2–5 as the wind strength increases up to 10 m s^{-1} . This influence is particularly important when the wind flows along the facade of the room of interest. In houses built with a crawl space the radon decrease may be as much as a factor of 100.

123. Diurnal and seasonal variations in houses were studied over the long-term [S18, D1, S32, D2, H5, J11, M7, S44]. Diurnal variations depend on the climate, occupancy and kind of ventilation (natural or forced ventilation) all of which affect the air exchange rate. In measurements in the United Kingdom [D1] the radon concentrations showed maxima during the night. In Finland [M20] the average values of all measurements in 37 dwellings showed maxima in the early morning and minima at noon, as shown in Figure XXIV. In

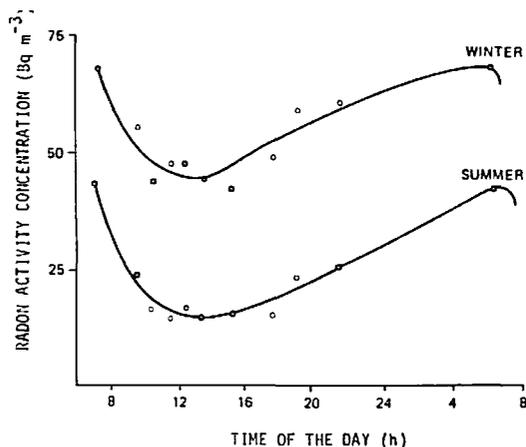


Figure XXIV. Diurnal variation of radon concentration in winter and summer [M20]

measurements made in the United States [S44] the maximum values were found in the morning and minimum at night. These diurnal variations occurred because of the use of cooling units during the nights [S44]. Variations by a factor of 10 or more between the minimum and maximum values were reported. However, the daily average values of the radon concentration during a week normally vary much less, within $\pm 50\%$ of the weekly average. These observations emphasize the need of continuous measurements or of several samples over at least 24 hours in order to obtain a representative value of the mean radon concentration in a house. If the radon concentration indoors is of the same order of magnitude as that outdoors, the same diurnal variations occur indoors as outdoors [H5]. The variations found illustrate the difficulties in obtaining a representative value for radon and thoron daughter exposure. Some of the uncertainty derives from the fact that only the levels occurring when the house is occupied determine the exposure, while levels at other times are irrelevant. This is discussed in [S44].

124. There are very few measurements of the seasonal variation of radon concentration in houses. In New York City during a period of two years (1972–1973), the seasonal variation amounted to a factor of 3 between a maximum in summer and a minimum in winter, i.e., the variations were within $\pm 50\%$ of the yearly average [F3]. A few measurements in New Jersey comparing the radon activity concentrations in the basement of a house in winter and summer indicate that 30% lower values occurred during the summer [S44]. In another study in Austria over six months, a minimum was obtained in May and a maximum in January. The variations of the monthly means were within $\pm 30\%$ of the overall average. The maximum in winter depended on inversion outdoors [S21].

125. If all relevant factors were known quantitatively, it would be possible to determine the radon concentration indirectly by measuring the exhalation rate and

the ventilation rate. However, the radon sources are not always easily identified and the ventilation rate may vary unpredictably due to human activities. Under controlled conditions it is possible to obtain good agreement between calculated and observed values [O6]. In 16 residential buildings in New Jersey and New York measurements were made, predominantly in cellars, of exhalation rate, ventilation rate and radon concentration [B10]. By taking the relevant factors into account, the correlation found between the calculated and measured radon concentrations was very good (correlation coefficient 0.94), as shown in Figure XXV [B10]. Figure XXVI shows the estimated variation of

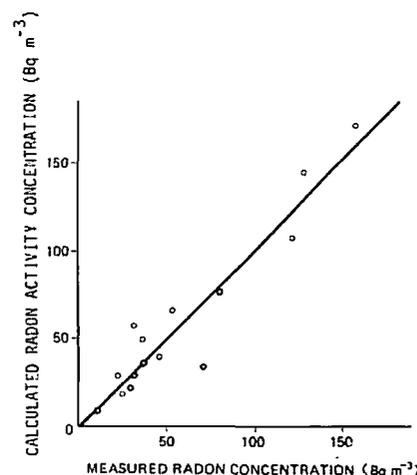


Figure XXV. Calculated radon activity concentration versus measured radon concentration in cellars [B10]

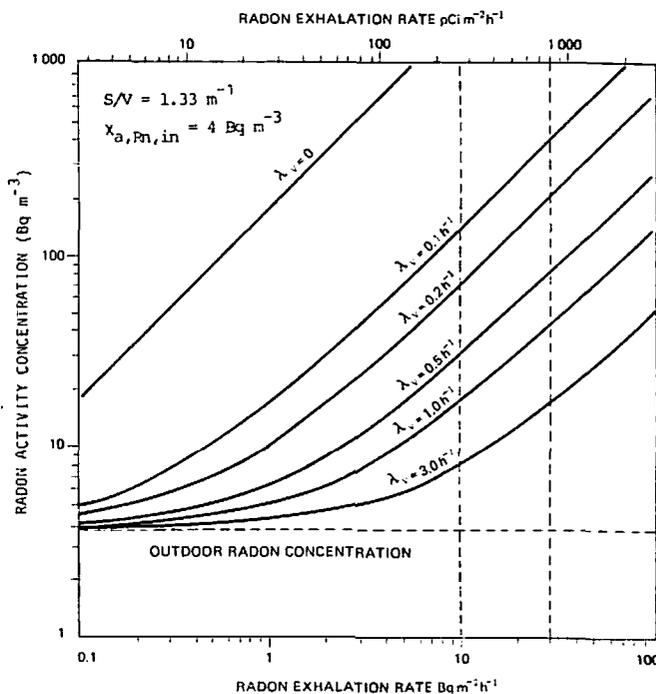


Figure XXVI. Radon activity concentration in room air as a function of the radon emanation rate for different values of the ventilation rate. The region marked with vertical broken lines corresponds to measured emanation rates from concrete [M17]

the radon activity concentration as a function of exhalation rate from building materials for various values of the ventilation rate [M17].

2. Tunnels

126. In principle, equation (23) for radon concentration in air is also applicable to mines and other underground spaces. However, owing to the great lengths of the tunnels in a mine, it is inappropriate to assume instantaneous mixing and homogeneous distribution of radon in the whole tunnel. Equation (23) may therefore be written in the form

$$\chi_{a, Rn}(x) = \frac{R p x + \dot{A}_k}{v} + \chi_{a, Rn, in} \quad (25)$$

where R is the radon exhalation rate per unit area in the mine; p is the peripheral length of a section through a mine tunnel; x is the distance along the mine tunnel; \dot{A}_k is the radon release rate for a source in the tunnel (e.g., radon-rich water); v is the volumetric air flow rate; and $\chi_{a, Rn, in}$ is the radon concentration in the inlet air (which may have passed other mine areas and which may have been contaminated with radon).

127. When \dot{A}_k and $\chi_{a, Rn, in}$ are small, the radon concentration increases continuously along the tunnel. However, the kind of ventilation system and the nature of the radon sources will greatly influence the level and changes of level of radon in a mine. Large radon release rates \dot{A}_k from uranium faces or radon-rich water will be rapidly diluted and transported along the tunnel or in ventilation tubes. Where radon-rich ventilation air $\chi_{a, Rn, in}$ from abandoned parts of a mine is allowed to enter active mining areas, it may give rise to an additional and rather constant radon concentration along the tunnels. The increase in potential alpha energy concentration of the radon daughters increases with the distance x along the tunnel from the ventilation point, and is proportional to $x^{1.85}$ [G24].

128. Diurnal and seasonal variations in radon concentration also occur in mines. The diurnal variation is especially pronounced at the time of the year when the temperature difference outdoors and in the mine changes sign over 24 hours. The resultant effect is a minimum during the night and a maximum during the day [S14]. The variation has been found to be $\pm 20\%$ of the daily average. The seasonal variations may be greater, $\pm 50\%$ of the yearly average or more, with a maximum during the summer and a minimum during the winter, as can be seen from Figure XXVII [S14].

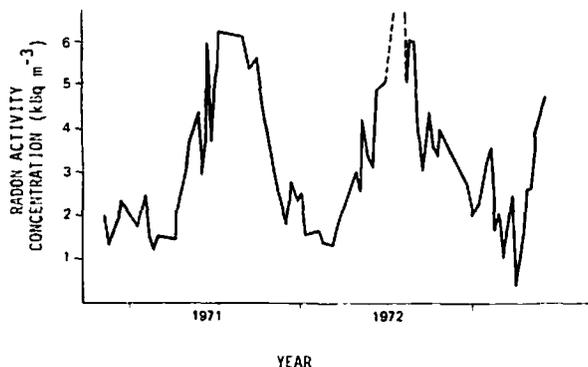


Figure XXVII. Radon activity concentration in the return air of Persberg Mine measured once a week [S14]

D. EQUILIBRIUM FACTOR F IN INDOOR AIR

129. A number of measured values of equilibrium factor F for radon daughters were given in Annex B of the 1977 report of UNSCEAR [U6]. No direct measurements on radon and radon daughters in houses with F values higher than 0.5 had been measured and the value chosen at that time was 0.5. Some new data have since been reported. In 21 New Jersey and New York residences the average fraction of unattached radon daughters was 0.07 and the F value for cellars was 0.52 ± 0.11 , for first and second floors 0.63 ± 0.14 , and for outdoor air 0.79 ± 0.11 [G1]. The difference between F values for cellar and upper floors is not significant, but, if correct, could be explained by assuming that the cellar is the main radon source for the upper floors as well, and that the ventilation of the cellar is mainly determined by ventilation of these floors. The radon concentration in the cellar was on average 73 ± 44 Bq m⁻³ and on the upper floors 34 ± 26 Bq m⁻³.

130. Measurements in apartments in the Federal Republic of Germany gave an average value of the equilibrium factor of 0.4 ± 0.1 . The results were based on 38 measurements and Figure XXVIII shows the distribution of the results [W14].

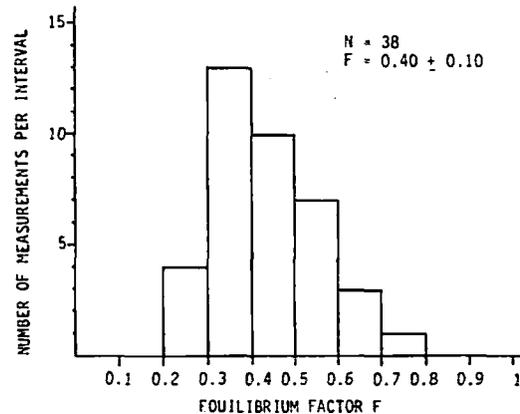


Figure XXVIII. Frequency distribution of equilibrium factor F for radon daughters in apartment rooms, based on 38 measurements [W14]

131. In an extensive investigation of the natural radiation in houses in Austria, radon and radon daughter activity concentrations were measured in 250 houses in Salzburg. The mean ratio $^{214}\text{Pb}/^{222}\text{Rn}$ was found to be 0.62 ± 0.08 , corresponding to a value of F of slightly less than 0.62 [S22].

132. Measurements in single-family houses in Sweden [S33] have given an average value of F in 3 houses of 0.45 ± 0.11 (range 0.28–0.65 in different rooms) with a mean radon concentration of 470 Bq m⁻³. The range of the distribution of F values in different rooms was narrower for each individual house than between houses and the standard deviation lay in the range 10–25%.

133. Another investigation concerned 12 single-family houses in Sweden [S34], 7 of which were built on waste piles from the milling of alum shale processing many years ago. With the exception of two houses, which were excluded because of extraordinary conditions, the average value of the equilibrium factor F was 0.60 ± 0.18 for the cellar and 0.60 ± 0.14 for the upper floor. In all houses the ventilation rate was measured by tracegas methods, and in Figure XXIX the values are

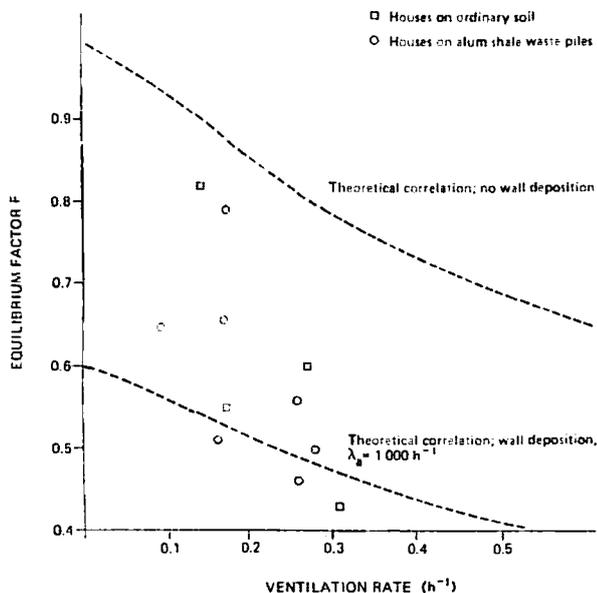


Figure XXIX. Equilibrium factor F in houses with different ventilation rates [S34]

plotted together with theoretical data taken from Table 5 and, also corrected for plateout, using a factor of 0.6 for $\lambda_a = 1000 \text{ h}^{-1}$ (see paragraph 34). There is no correlation between F and the ventilation rate in Figure XXIX and the F values are often higher than expected even for a rather dusty atmosphere. However, all values are less than predicted from Table 5, with no correction for wall deposition. Even if it is difficult to explain the high F values on a theoretical basis, there is experimental evidence that F values higher than those predicted do occur and are often higher than 0.5.

134. Other measurements, however, indicate lower values of F . In March Township, Ontario, Canada [L7] an average equilibrium factor of 0.38 was obtained. In measurements in 25 dwellings in Norway [S24] the mean value of F was 0.5 with a range of 0.3–0.8; in 35 dwellings in Finland [M20] the mean value of F was 0.47, with a range of 0.30–0.63 (Table 17), and in the United Kingdom a mean value of $F = 0.5 \pm 0.2$ was observed from measurements in 200 houses [O7].

135. The equilibrium factor in mines is highly dependent on the ventilation rate and the distance from the radon source. It can be estimated from published data on radon and radon daughter concentrations. The results from 60 measurements in uranium mines in New Mexico and Colorado, United States [R3], corrected by Kotrappa and Mayya [K12], give an F value of 0.29 ± 0.16 . Another compilation of several measurements in uranium mines in the United States gives an average value of F of 0.32 [H21] and in uranium mines in France an average value of F of 0.17 is used [P17]. In non-uranium mines average F values of 0.5–0.6 in Norway [M19, S54], 0.7 ± 0.1 in Sweden [S15], 0.7 in non-coal mines in the United Kingdom [S52] and 0.3 in Poland [D15] have been found.

136. In view of the variations in the F values found in the measurements there is no good reason to change the average value of 0.5 adopted in Annex B of the UNSCEAR 1977 report [U6] for houses, bearing in mind that this value may lead to an underestimate of the equilibrium equivalent concentration in poorly ventilated houses. For uranium mines an F value of 0.3 is probably applicable while for non-uranium mines an

F value of 0.7 may be used. For many non-uranium mines the ventilation is good and F is lower than 0.7. In poorly ventilated mines however the value of F is likely to be higher than those given above.

IV. EXPOSURE-DOSE RELATIONSHIPS

A. INHALATION

137. The inhalation of ^{220}Rn —or ^{222}Rn —gas itself leads to a rather uniform distribution of these noble gases in the whole body. Due to the low solubility of these inert gases in body tissues the resulting effective dose equivalent from the inhalation of radon gas itself is normally small compared with the radiation dose from inhaled radon daughters; only in ore bodies with a very low equilibrium factor does the contribution from radon have to be taken into account. The following discussion is therefore concerned mainly with the estimation of the relationship between the exposure or potential α -energy intake of short-lived radon daughters by inhalation and the resulting dose in target tissues of the lung.

138. As mentioned in Annex B of the 1977 report [U6], in the special case of inhaled short-lived radon daughters the dose to the bronchial epithelium is considerably higher than the dose to the pulmonary tissue or the mean dose to the total lung. The cells at risk from the α radiation by radon daughters which are deposited on the surface of the bronchial airways and transported upwards in the bronchial tree by the mucociliary escalator are assumed to be located in the basal cell layer of the bronchial epithelium [I13]. Therefore the lung dosimetry for inhaled radon daughters has to consider two target tissues in the lung: the tracheobronchial basal cell layer (T-B) and the pulmonary epithelium (P), the latter including the alveolar region and the non-ciliated terminal bronchioles. In the cases of the longer-lived ^{222}Rn daughters ^{210}Pb and ^{210}Po , and of the ^{220}Rn daughter ^{212}Pb , a considerable fraction of the activity deposited in the respiratory tract is transferred to tissues other than the lung, particularly to bone and kidney. The dose to these other tissues can be evaluated by applying the dose assessment models for internal exposure described in Annex A.

139. The analysis of the dose distribution in the respiratory tract from inhaled radon daughters proceeds from the calculation of the equilibrium activity distribution, taking into account the deposition pattern, the mucociliary clearance in the bronchial tree, the possible uptake and retention in epithelial tissues and the transfer to blood. An important parameter for the evaluation of the dose to the bronchial basal cell layer is the depth distribution of this cell layer throughout the bronchial tree. The limited knowledge about this depth distribution has been one of the main sources of uncertainty in previous dosimetric models. Information now available, particularly the experimental data from Gastineau et al. [G25], offers the possibility of reducing this uncertainty. In addition, more realistic data about the fraction of unattached daughter atoms and about the activity median diameter (AMD) of the aerosol which carries the daughter atoms are now available, as outlined in the previous chapters. Together with improved models of the anatomical structure of the bronchial tree, this information allows a more realistic estimate of the exposure-dose relationships for the target tissues in the lung.

140. On the basis of this knowledge improved dosimetric models for the inhalation of short-lived radon daughters have been developed since the publication of the 1977 report [U6]. Of main importance are the studies of Harley et al. [H24], Hofmann et al. [H12], Jacobi et al. [J13, J18] and James et al. [J19]. The results of these studies have also been used by the ICRP for the assessment of occupational limits for the inhalation of ^{222}Rn , ^{220}Rn and their short-lived daughters on the basis of a dosimetric approach [I13]. A comprehensive summary and comparison of these dosimetric models and their results has been prepared by an expert group of the NEA/OECD (see [J19]). In this report a sensitivity analysis was also performed showing the influence of the various physical and biological parameters on the dose to the target tissues in the lung.

141. All of these new studies agree that under typical conditions in mines, as well as in houses, the dose distribution over the basal cell layer of the bronchial generations from inhaled radon daughters is not so inhomogeneous as was previously assumed but shows a rather broad maximum in the region from the lobar bronchi down to the upper bronchioles. This conclusion is strengthened by the fact that each study used different models for the anatomical lung structure, for the deposition and retention of radon daughters, and for the depth distribution of the basal cells. Only in the case of abnormally high values of the unattached fraction of the total potential alpha energy of the daughter mixture should a significant peak in the dose distribution be expected in the segmental bronchi, and this only if an increased deposition of unattached atoms due to turbulent air streaming in the upper bronchi is assumed. On the basis of these findings a mean dose to the bronchial basal cell layer can be derived, averaged over the above mentioned bronchial generations.

142. From the dosimetric models it follows that the conversion factors between the inhaled potential alpha-energy intake I_{pot} of the daughter mixture and the committed dose equivalent H to the bronchial basal cell layer and to the pulmonary epithelium shows a nearly linear relation with the unattached fraction f_p of the total potential alpha energy of the daughter mixture

$$H_{T-B}/I_{\text{pot}} = a_{T-B} + b_{T-B} f_p \quad (26)$$

$$H_P/I_{\text{pot}} = a_P (1 - f_p) \quad (27)$$

The first term a in these equations gives the conversion coefficient for the attached daughters, which depends on the activity median diameter (AMD) of the carrier aerosol. The influence of breathing rate on the quotient of the dose equivalent to the intake is relatively small. It is of great practical importance that these linear relationships are nearly independent of the equilibrium of the daughter nuclides ratios and therefore of the corresponding equilibrium factor F .

143. The available measurements in mine areas as well as in houses indicate that the unattached fraction f_p of the total potential α -energy of ^{222}Rn daughter mixtures is relatively small. Averaged over long periods of time f_p is equal to about 0.05 (see paragraph 20). Under these conditions with an AMD = 0.2 μm , the different dosimetric models give a mean dose equivalent to the bronchial basal cell layer per unit of

inhaled potential alpha-energy intake of ^{222}Rn daughters of

$$(H_{T-B}/I_{\text{pot}})_{^{222}\text{Rn daughters}} = 15 - 40 \text{ Sv J}^{-1}$$

where a quality factor $Q = 20$ is applied for alpha radiation. For areas with relatively high dust concentration a value in the lower part of this range should be expected, whereas for highly-ventilated areas with low dust concentration a value in the upper half of this range might be more appropriate. For the pulmonary epithelium, the dose equivalent per unit potential alpha-energy intake is

$$(H_P/I_{\text{pot}})_{^{222}\text{Rn daughters}} = 2 - 5 \text{ Sv J}^{-1}$$

Thus, on average, the dose to the bronchial basal cell layer is about a factor of 5-8 higher than the dose to the pulmonary region. As mentioned earlier, the variation with breathing rate of these dose equivalents per unit intake is relatively small and falls within the quoted range. For an AMD of 0.1 μm , which might be typical for outdoor air, the dose equivalents per unit intake are about a factor of 1.5 higher than the values given above.

144. In the case of inhaled ^{222}Rn daughters the dose to tissues other than the lung is rather small and can be neglected in the evaluation of the effective dose equivalent. For the evaluation of the weighted lung dose equivalent a weighting factor $w = 0.12$ is usually applied to the mean lung dose equivalent, when the total lung is considered as target tissue (see Annex A). For the special case of inhaled radon daughters this "mean lung dose" concept might not be appropriate, because of the considerably higher dose to the bronchial epithelium. Following the recommendations of ICRP [I13], it might be reasonable to split the weighting factor for the total lung and to apply a weighting factor $w = 0.06$ to each of the two target tissues in the lung. From the values of the dose equivalents per unit intake given in paragraph 143, the following figures for the effective dose equivalent per unit inhaled potential alpha energy of ^{222}Rn daughters can be derived:

$$\text{Mean lung dose concept} \quad H_{\text{eff}}/I_{\text{pot}} = 0.3 - 0.8 \text{ Sv J}^{-1} \\ (w_{\text{lung}} = 0.12)$$

$$\text{Regional lung dose concept} \quad H_{\text{eff}}/I_{\text{pot}} = 1 - 3 \text{ Sv J}^{-1} \\ (w_{T-B} = w_P = 0.06)$$

These values refer to an AMD = 0.2 μm and can be applied for the occupational exposure in mines and for the indoor exposure in houses. In this report a reference value of $(H_{\text{eff}}/I_{\text{pot}})_{^{222}\text{Rn daughters}} = 2 \text{ Sv J}^{-1}$ is used, taking into account the high dose contribution to the bronchial basal cell layer. It should be noted that the ICRP recommends an effective dose equivalent per unit inhaled potential alpha-energy intake of ^{222}Rn daughters of 2.5 Sv J^{-1} for the radiation protection of workers in mines [I13]. This value was derived on the basis of a comparison of the dosimetric approach with the observed excess lung cancer risk of radon-exposed miners, particularly uranium miners. For the exposure in outdoor air a value of 3 Sv J^{-1} is used for ^{222}Rn daughters in this report.

145. Dose equivalents per unit intake of inhaled ^{220}Rn daughters can be derived in a similar way from the new

dosimetric models [J13, J19]. Table 18 shows the resulting dose equivalent in the target tissues of the lung and in other relevant tissues per unit of inhaled potential alpha energy, separately for ^{212}Pb and ^{212}Bi , which are the most important daughter nuclides of ^{220}Rn . The values refer to an AMD of 0.2–0.3 μm . The dose range given is based on a transfer half-life time from the lung to blood of 0.2–0.5 days, as it follows from human inhalation studies with ^{212}Pb [H18]. The dose equivalent per unit intake for tissues other than the lung were evaluated by applying the dosimetric models recommended by ICRP [I10] (see also Annex A). It follows from Table 18 that, in addition to the dose equivalent to the lung, the dose equivalents to bone surfaces, kidney and liver have to be taken into account. With respect to the effective dose equivalent the difference between the two weighting concepts for the lung is small. The potential alpha energy per unit activity inhaled is about 10 times higher for ^{212}Pb than for ^{212}Bi . This means that if a mixture of both radionuclides is inhaled the contribution to the dose equivalent from the inhaled ^{212}Pb is dominating. In this report a reference value

$$(H_{\text{eff}}/I_{\text{pot}})_{^{212}\text{Pb}+^{212}\text{Bi}} = 0.7 \text{ Sv J}^{-1}$$

is used for the evaluation of the effective dose equivalent per unit inhaled potential alpha energy of ^{220}Rn daughters. This value is applied to the occupational exposure in mines as well as to the natural exposure from these nuclides. The ICRP recommends an effective dose equivalent per unit inhaled potential alpha-energy intake of 0.8 Sv J⁻¹ for occupational exposure to these nuclides [I13]. It should be noted that the effective dose equivalent per unit of inhaled potential alpha energy of these ^{220}Rn daughters is about one-third of the corresponding value for short-lived ^{222}Rn daughters. Compared with $^{212}\text{Pb}+^{212}\text{Bi}$ the contribution from inhaled ^{220}Rn to the effective dose equivalent is normally small. For the inhalation of $^{220}\text{Rn}+^{216}\text{Po}$ an effective dose equivalent per unit activity of inhaled ^{220}Rn of about $1 \cdot 10^{-10}$ Sv Bq⁻¹ has been estimated [J13], where the main contribution results from the alpha emission of these nuclides in the air volume of the lung.

146. The potential alpha-energy intake I_{pot} by inhalation during a time period T is given by the relationship

$$I_{\text{pot}}(T) = V(T) C_{\text{pot}} = \dot{I}_{\text{ih}} T C_{\text{pot}} = \dot{I}_{\text{ih}} \check{C}_{\text{pot}} \quad (28)$$

where C_{pot} is the mean potential alpha energy concentration in air and V the total breathing volume during the time period T . The inhaled air volume V is the product of T with the mean breathing rate \dot{I}_{ih} during this period. The product $\check{C}_{\text{pot}} = C_{\text{pot}} T$ is called the potential alpha-energy exposure during the period T . On the basis of this relationship, conversion coefficients between alpha-energy exposure C_{pot} and dose equivalent can be derived from the given dose equivalents per unit intake. The SI unit for the potential alpha-energy exposure is J s m⁻³ or J h m⁻³. Older units like WLM or WL h are defined in paragraphs 12 to 16, where the conversion coefficients between the potential alpha-energy concentration and the equivalent equilibrium concentration (EEC) of radon are also given.

147. For the occupational exposure of miners a mean breathing rate $\dot{I}_{\text{ih}} = 20 \text{ l min}^{-1} = 1.2 \text{ m}^3 \text{ h}^{-1}$ is assumed

[I13]. Taking into account a working period $T = 2000$ hours per year, this yields an inhaled air volume of $V = 2400 \text{ m}^3$ during this period. In Table 19 the resulting conversion coefficients between potential alpha-energy intake \dot{I}_{ih} or the radon daughter exposure \check{C}_{pot} , respectively, and the effective dose equivalent are summarized for occupational conditions in mines.

148. For the evaluation of the population exposure by inhalation of radon daughters in houses an average occupancy factor of 0.8 was assumed. This corresponds to an average residence time of

$$T_{\text{in}} = 19 \text{ h d}^{-1} = 6935 \text{ h a}^{-1}$$

indoors, and

$$T_{\text{out}} = 5 \text{ h d}^{-1} = 1825 \text{ h a}^{-1}$$

outdoors. In Table 20 the assumed breathing rates during the indoor and outdoor residence periods are given. The corresponding average breathing rates are as follows:

$$\dot{I}_{\text{ih, in}} = 0.8 \text{ m}^3 \text{ h}^{-1} = 15 \text{ m}^3 \text{ d}^{-1} = 5475 \text{ m}^3 \text{ a}^{-1}$$

during the indoor residence period, and

$$\dot{I}_{\text{ih, out}} = 1 \text{ m}^3 \text{ h}^{-1} = 5 \text{ m}^3 \text{ d}^{-1} = 1825 \text{ m}^3 \text{ a}^{-1}$$

during the outdoor residence period. The average volumes of air inhaled in a year are thus $V_{\text{in}} = 5475 \text{ m}^3$ and $V_{\text{out}} = 1825 \text{ m}^3$ during the indoor and outdoor residence periods, respectively.

149. The annual inhaled potential alpha-energy intake of radon daughters is

$$I_{\text{pot, in}} = (V C_{\text{pot}})_{\text{in}} = (\dot{I}_{\text{ih}} T C_{\text{pot}})_{\text{in}} = (\dot{I}_{\text{ih}} \check{C}_{\text{pot}})_{\text{in}} \quad (29)$$

for the indoor residence period, and

$$I_{\text{pot, out}} = (V C_{\text{pot}})_{\text{out}} = (\dot{I}_{\text{ih}} T C_{\text{pot}})_{\text{out}} = (\dot{I}_{\text{ih}} \check{C}_{\text{pot}})_{\text{out}} \quad (30)$$

for the outdoor residence period. \check{C}_{pot} is the radon daughter exposure, which is the time integral of the radon daughter concentration during the time period considered. In Table 21 the reference values for the effective dose equivalent per unit of inhaled potential alpha energy and per unit of radon daughter exposure are summarized. These values are used in this report for the evaluation of the population dose from radon daughters indoors and outdoors.

150. For practical purposes it is reasonable to give direct conversion coefficients between the concentrations indoors and outdoors and the expected annual effective dose equivalent. The quotients of the annual effective dose equivalent to the potential alpha-energy concentrations are given by the relationships

$$\frac{H_{\text{eff, in}}}{C_{\text{pot, in}}} = (H_{\text{eff}}/I_{\text{pot}})_{\text{in}} V_{\text{in}} \quad (31)$$

$$\frac{H_{\text{eff, out}}}{C_{\text{pot, out}}} = (H_{\text{eff}}/I_{\text{pot}})_{\text{out}} V_{\text{out}} \quad (32)$$

where $H_{\text{eff}}/I_{\text{pot}}$ are the effective dose equivalents per unit potential alpha-energy intake, given in Table 21; and V_{in} and V_{out} are the annual inhaled air volumes indoors and outdoors, as given in paragraph 148. The values of $H_{\text{eff}}/C_{\text{pot}}$ are presented in Table 22. Taking into account the relative volumes of air breathed indoors and outdoors, the annual effective dose equivalent per unit inhaled activity of radon daughters is estimated to be $1.3 \cdot 10^{-8}$ Sv Bq⁻¹.

151. The reference values for the dosimetric coefficients given in Tables 21 and 22 refer to adult members of the public. Correction factors should be applied for infants and children, taking into account the change of lung mass and breathing rate with age. An age-dependent lung model has been developed by Hofmann et al. [H12] and applied to the inhalation of radon daughters. This model indicates that for a given radon daughter concentration in air the effective dose equivalent probably reaches a maximum value at the age of about six years which is about 2.5 times higher than the effective dose equivalent at the age of 30 years. On the average, for the age group up to 10 years the effective dose equivalent might be about a factor 1.5–2 higher than for adults [J19]. This correction factor refers to the quotients of effective dose equivalent to exposure given in Table 21.

B. INGESTION

152. Ingestion of water containing dissolved radon results in a radiation dose to the body from the radon gas and the radon daughters in the water. The main part of the ingested radon is eliminated from the body very rapidly through the lungs. The resultant dose equivalent has been estimated by several investigators on the basis of experimental studies of exhaled radon following the ingestion of radon-rich water [A13, B23, B24, H25, K23, S17]. The stomach, whole body, liver, kidneys, fat and marrow have been considered as target organs. The largest dose is estimated to be received by the stomach.

153. For the stomach the absorbed dose per unit activity of ²²²Rn ingested varies between about 50 and 200 nSv Bq⁻¹. Using a weighting factor of 0.06, the effective dose equivalent per unit activity of ²²²Rn ingested is thus 3–12 nSv Bq⁻¹. In general, most of the daily intake of water is ingested in food and beverages such as tea and coffee, rather than by direct consumption. Much of the dissolved radon is released from water during cooking and boiling and the only significant radon intake comes therefore from the drinking of water itself. The amount of water consumed in this way varies between 0.3 and 1.2 l d⁻¹ [S17]. Assuming a consumption of 0.5 l d⁻¹, and an effective dose equivalent per unit activity ingested of 3 nSv Bq⁻¹, a radon concentration of 1 kBq l⁻¹ leads to an annual effective dose equivalent of 0.5 mSv (range 0.3–5 mSv).

V. LEVELS AND DOSES

A. RADON AND THORON IN OUTDOOR AIR

154. The concentrations of radon and thoron and their daughters in air vary with place, time, height above ground and meteorological conditions. Estimates of annual averages at different locations requires frequent sampling over the long term and such measurements are rare. Table 23 summarizes data for radon and ²¹²Pb concentrations at various geographic locations, as given in the references or calculated from published data. Some estimates of radon daughters assume radioactive equilibrium with radon, and this is seldom the case. The equilibrium factor *F* for radon daughters (as defined in paragraph 15) is assumed, as in Annex B of the 1977 report [U6], to be 0.6. The value of *F* in outdoor air depends on meteorological parameters and lower values of *F* (0.4–0.6) are not uncommon. The average value of the ratio of the thoron daughter ²¹²Pb to ²²²Rn is observed to be 0.06 ± 0.04 , and implies a thoron daughter activity concentration which is about a factor of 10 lower than for radon daughters.

155. Since the values in Table 23 are not necessarily representative of the locations concerned, they are not easily comparable. However, the average values for continental, island and ocean air probably are representative. These are 3 Bq m⁻³, 0.1 Bq m⁻³ and 0.1 Bq m⁻³, respectively. For the estimation of average dose equivalent, a radon concentration in air over land of 3 Bq m⁻³ might be used, corresponding to an equilibrium equivalent concentration of 1.8 Bq m⁻³. Using the quotient of effective dose equivalent per unit equilibrium equivalent concentration given in Table 22, the corresponding average annual effective dose equivalent from radon daughters is $5.6 \cdot 10^{-2}$ mSv, assuming that 5 hours are spent each day outdoors. Using a ²¹²Pb/²²²Rn ratio of 0.06, the corresponding average annual effective dose equivalent from thoron daughters (assuming equilibrium between ²¹²Pb and ²¹²Bi) is $1.8 \cdot 10^{-2}$ mSv.

156. Higher radon and radon daughter concentrations in outdoor air are likely in the vicinity of coal-fired and geothermal power stations and uranium tailings. Table 24 presents normalized annual effective dose equivalents from outdoor air for such sources, calculated from equations (19) and (20) using the dose conversion factors given in Table 22. An equilibrium factor of 0.6 and a value of 0.2 for the fraction of time spent outdoors were assumed. The radon dispersion coefficients used are illustrated in Figure XVIII. Normalized release rates of 60 GBq (GW a)⁻¹ and 370 TBq (GW a)⁻¹ were used for coal-fired and geothermal stations, respectively, and a stack height of 100 m was assumed. It is not easy to provide a clear-cut figure for the normalized release from tailings because it depends on their configuration and surface area. However, taking the figure of 10⁹ Bq (MW a)⁻¹ (Annex F, Table 2) from an area of 0.6 hectare (or an exhalation rate of 5.3 Bq m⁻² s⁻¹). Table 24 gives normalized annual effective dose equivalents for tailings areas of 10⁴ and 10⁶ m², respectively. To estimate the total radiological impact, the values in Table 24 should be increased by about a factor of 3 to include the effects of indoor exposure.

B. RADON IN WATER

157. Measurements of radon in water have mostly been associated with the study of correlations between uranium/radium concentration in rock and radon concentration in ground water. In general, they have

been undertaken only in uranium-bearing areas, or regions with a known high concentration of radon in the water. Measurements intended to estimate the weighted average radon concentration in water for a country or community are rare, because radon in water has not been considered to be a significant factor for human exposures, except in areas with very high radon levels. However, the recent interest in radon exposure in houses has indicated radon in water to be a potentially significant source in some cases. The results of some measurements of radon in water are shown in Table 25 and more extensive data may be expected in the future. It must be emphasized that surface water in many countries is the mean source of drinking water and this water is normally expected to have low radon concentration.

158. The data presented by Hess [H10] for the United States are probably not representative, and perhaps not even for the counties examined. Most of the wells were drilled artificially and selected with special emphasis on granite and pegmatite areas where high radon concentrations were expected. The measurements follow approximately a log-normal distribution. The results from North Carolina [S1] include only those water samples with statistically significant activity concentrations of radon ($> 700 \text{ Bq m}^{-3}$), without mention of the number of values discarded. Other studies of radon concentrations in the United States have been reported for water samples from North Carolina [A1], Western United States [O1], Hot Springs, Arkansas [K15], Great Salt Lake, Utah [T2] and South Texas [G7]. The results were presented as examples of high values of radon in water supplies. Of these water sources 74% had a radon concentration of less than 74 kBq m^{-3} and 5% above 370 kBq m^{-3} [D14].

159. The results from Finland in Table 25 refer to 192 drilled wells. These include recent results together with other extensive measurements [A7, C4, A11, K18]. A summary of data is given in Table 26 and in Figure XXX. The results are found to follow a log-normal

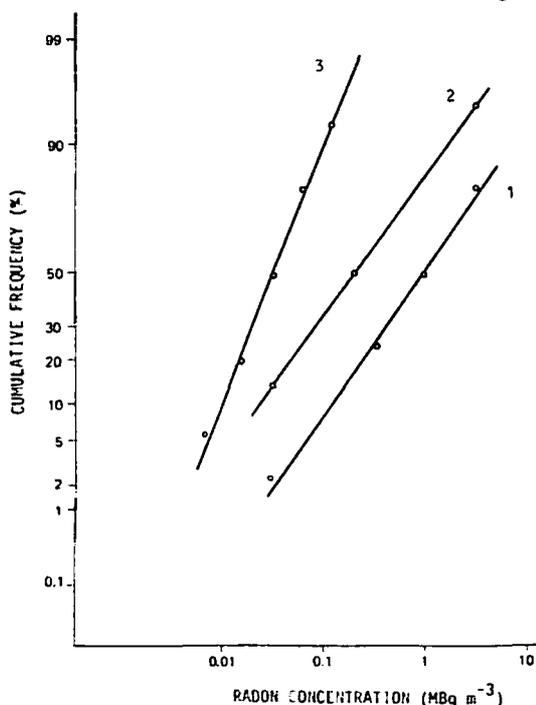


Figure XXX. The distribution of radon concentrations in (1) bored wells in the Helsinki area; (2) bored wells in Finland outside Helsinki; (3) other wells in Finland (excluding waterworks) [A7, C4]

distribution. The population-weighted average is 56 kBq m^{-3} , which must be considered as a relatively high value.

160. The Swedish data represent water from almost all waterworks, delivering more than 10^5 m^3 of ground water per year to about 3 million people. In addition, 3.7 million people in Sweden use surface water from waterworks and the rest, about 1 million people, use private wells. If the weighted average of the radon concentration in private wells is assumed to be $10\text{--}100 \text{ kBq m}^{-3}$ [S11], the weighted national average is $10\text{--}20 \text{ kBq m}^{-3}$.

161. In the United Kingdom natural radon levels in water supplies are lower than in many other countries [H9]. Two extensive surveys carried out in the early 1960s [T5, K6] found the highest radon concentration (26 kBq m^{-3}) in a reservoir on granite. However, in highly mineralized granite areas (Cornwall and Devon) the radon concentration in some water sources used for supply is about 700 kBq m^{-3} . Various spa waters have up to 70 kBq m^{-3} .

162. A report from the Federal Republic of Germany summarizes the concentrations of natural radionuclides in water, food and in man [M18]. For radon in drinking water the average values lie in the range of $0.4\text{--}4 \text{ kBq m}^{-3}$.

163. In conclusion, available measurements on radon in water cannot easily be used to estimate weighted mean values for a country as a whole. The measurements often refer to areas of special geological interest, because of their high content of uranium or radium or of special radiological interest because of their high radon levels. It seems likely that only a small proportion of the world's population, perhaps between 1% and 10%, consumes water containing concentrations of radon of the order of 100 kBq m^{-3} or higher, drawn from relatively deep wells. For the remainder who consume water from wells or surface sources, the weighted world average concentration from all sources is probably less than 1 kBq m^{-3} . A small proportion ($< 1\%$) consumes water containing $1\text{--}10 \text{ MBq m}^{-3}$ of radon, and there have been a few exceptional reports of concentrations in the range $10\text{--}100 \text{ MBq m}^{-3}$.

164. The radiation dose caused by radon in water is due partly to ingestion and partly to inhalation of the radon daughters produced by decay of the radon. The relevant annual effective dose equivalents from water containing 1 MBq m^{-3} of radon are 0.5 mSv (ingestion), and 6 mSv (inhalation of radon daughters). The latter value can be calculated from the relationships given in paragraphs 96 and 136 and in Table 22. If one assumes a value of 3 kBq m^{-3} for the average radon concentration in water, the corresponding annual effective dose equivalent is about $20 \text{ } \mu\text{Sv}$.

C. RADON IN HOUSES

165. There is at present considerable interest in this area, particularly in the light of domestic energy conservation programmes. Radon concentrations in the domestic environment are also being affected by the use of land fill and of building materials which have radium concentrations significantly above normal. The following paragraphs summarize available data but it is expected that many new data will emerge in the literature from current studies.

(a) Austria

166. Extensive measurements of radon, thoron and their decay products have been made in Salzburg, Austria [S22]. To obtain representative values a number of test families and individuals were assessed (729 individuals in all). Special efforts were made to correct for time-variations in the activity concentrations. The values of radon concentrations were observed to be log-normally distributed. The mean concentrations were found to be 22 Bq m^{-3} for radon, 12 Bq m^{-3} for radon daughters (an equilibrium factor of 0.56 was used, based on reported mean radon/daughter ratios) and 1 Bq m^{-3} for ^{212}Pb . Maximum concentrations were more than a factor of 10 higher than the average. The reported radon daughter concentrations correspond to an annual effective dose equivalent of 0.7 mSv.

(b) Canada

167. An extensive survey was recently reported from Canada [M6, M26]. It concerned a total of 13 436

houses selected at random in 19 Canadian cities, and measured in 1977, 1978 and 1980. The sample corresponded to about 0.4% of the houses in the cities studied. Basements (or ground floors in houses with no basement) were the preferred sampling locations. The houses were regarded as typical for Canada, made with wood frames and little stone. Interior wall linings of painted natural gypsum were common and were found to be an insignificant source of radon. Results are summarized in Table 27. The average equilibrium equivalent concentration of radon is 17 Bq m^{-3} , corresponding to an annual effective dose equivalent of 1 mSv. Since the measurements were performed in summer periods, the annual average might in fact be higher. Radon appeared to originate from radium in nearby soil. Geographical differences were statistically significant. Variations within cities were however generally greater than variations between cities. The equilibrium factor calculated from the geometric means is on the average 0.52 ± 0.12 . The results of all measurements show a log-normal distribution, which is illustrated in Figure XXXI.

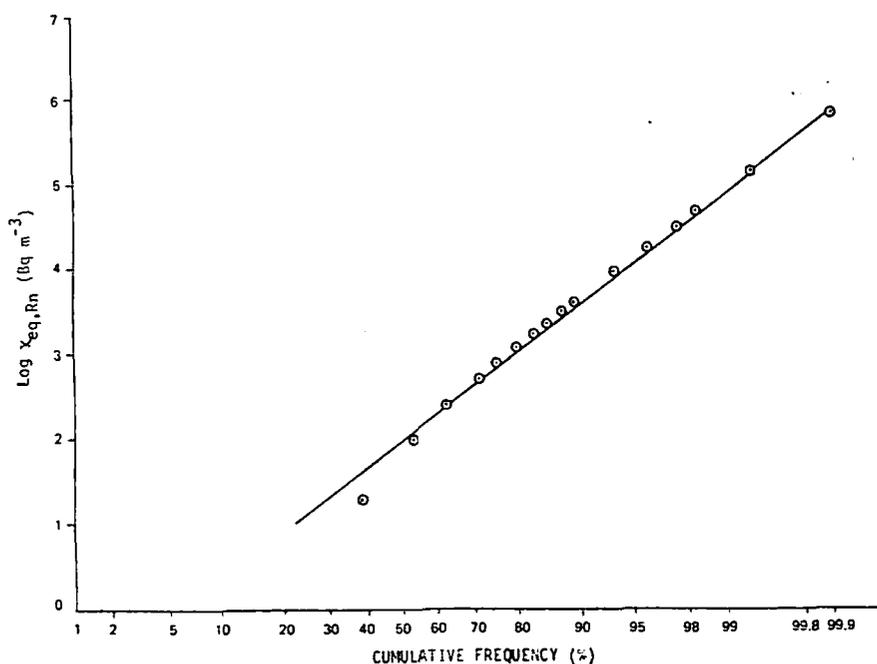


Figure XXXI. Log-normal cumulative frequency plot of radon daughter concentrations in Canadian homes (1977, 1978, 1980 surveys)

168. In March Township, Ontario, Canada, 343 houses were surveyed because of the discovery of nearby low-grade uranium deposits (5 ppm of uranium). Based on radon daughter measurements, 56.6% of the houses had equilibrium equivalent concentrations of radon lower than 19 Bq m^{-3} . The highest value was 700 Bq m^{-3} and the arithmetic average was 50 Bq m^{-3} [L7, T8].

169. The area of Castlegar-Trail in British Columbia, Canada, has been surveyed because of high radon concentrations in water. Based on radon daughter measurements, 46.2% of the homes had equilibrium equivalent concentrations lower than 19 Bq m^{-3} ; the highest value observed was 2900 Bq m^{-3} and the average was 74 Bq m^{-3} [L7].

170. Other investigations have been made in Canada [K9] in areas where radioactive contamination was suspected for various reasons in radium-uranium refineries, metallurgical operations making use of refinery

residues and uranium mining. One area was classified as a "normal" reference area. The results are shown in Table 28.

171. The influence of radon in water on the levels of radon and radon daughters in houses was studied in a survey in Canada including 6 houses, 5 trailer-homes and 2 schools. The water supplies originated from 15–62 m deep wells in granite. The results are seen in Table 29 [M7]. The air-to-water concentration ratio (see paragraph 96) is small (average $2 \cdot 10^{-5}$), probably because of good ventilation. The low equilibrium factors F support such a conclusion. Table 30 shows the resulting increase of radon concentration in air due to a warm water shower with water containing 4.4 kBq m^{-3} radon [M7]. The decrease of the radon concentration after the shower corresponds to a ventilation rate of $1\text{--}3 \text{ h}^{-1}$ and the radon daughter concentration increases only to about 10% of the equilibrium value. The air-to-water concentration ratio is $(2\text{--}8) \cdot 10^{-4}$ in this case.

(c) Finland

172. Between 1977 and 1978 a preliminary survey on radon and radon daughter concentrations in 35 normal Finnish dwellings was carried out in the Helsinki area [M20]. Buildings of different ages, types and building materials were included. The radon concentration in tap water was below 100 kBq m^{-3} and therefore it was concluded that the main sources of radon were the building materials and the soil. The mean equilibrium equivalent concentration of radon during the day was 13 Bq m^{-3} and the radon concentration 27 Bq m^{-3} . The 24 h mean value of radon concentration was 44 Bq m^{-3} and the equilibrium equivalent concentration of radon 17 Bq m^{-3} , corresponding to an annual effective dose equivalent of 1 mSv. The survey included too many concrete houses and too few wooden ones for the mean to be regarded as representative of radon concentrations in Finnish dwellings. Later measurements in the same area with integrating-type instruments gave higher mean concentrations. This is believed to be due at least in part to higher concentrations during the night than those estimated in the first series of measurements. The highest concentrations (up to the level $10\,000 \text{ Bq m}^{-3}$) were found in houses where the only contributor can be radon from the ground or bedrock.

173. Radon in water may be a significant source of radon to air in dwellings in many countries. In the region of Helsinki, Finland, very high radon concentrations in water have been reported [C3, K1, K2]. The highest concentration measured was 44 MBq m^{-3} . In a recent study, radon measurements in air were carried out in 20 houses with radon concentration in water between 150 kBq m^{-3} and 17 MBq m^{-3} . The rooms in a house were classified according to the use of water. The results are given in Table 31 [A3].

174. The table shows that the radon concentrations were sometimes remarkably high and, as expected, higher in rooms and situations where water was used (bathrooms, etc.). Under such conditions it is expected that the equilibrium factor F is smaller than average. The air-to-water concentration ratio was found to be the following in the three groups of rooms: $(21 \pm 9) 10^{-4}$ in wet rooms; $(6 \pm 3) 10^{-4}$ in ordinary rooms with water; $(0.6 \pm 0.3) 10^{-4}$ in living rooms, etc. (90% confidence level). A weighted average value would be $1.4 10^{-4}$, assuming 0.5 h spent in wet rooms, 2 h in ordinary rooms with water and 17 h in other rooms. Using an equilibrium factor F of 0.5 (which may be a slight over-estimate), and average values for the radon concentration in water and for the air-to-water concentration ratio, the weighted equilibrium equivalent radon concentration is 360 Bq m^{-3} . The maximum value would amount to 1200 Bq m^{-3} . The results of a follow-up study [A4] are within the range of the above-mentioned values. The air-to-water concentration ratios are higher and much more variable for wet rooms (10^{-4} to 10^{-2}) than for living rooms. Measurements of radon daughters by track-etch dosimeters carried by persons during three-day periods give a ratio of equilibrium equivalent radon concentration to radon concentration in water of $(0.4 \pm 0.3) 10^{-4}$.

(d) Germany, Federal Republic of

175. Measurements in the Federal Republic of Germany have recently been published [W14]. The study covers 32 houses, grouped according to their principal construction material; equilibrium equivalent radon and thoron concentrations, normalized to a

ventilation rate of 0.5 h^{-1} are estimated from the measured exhalation rates. The results are summarized in Table 32. Mean values for radon and thoron are 8.1 Bq m^{-3} and 0.37 Bq m^{-3} , respectively, corresponding to annual effective dose equivalents of 0.5 mSv and 0.1 mSv. Other measurements [J15] on about 250 dwellings indicate a log-normal distribution with geometric means equilibrium equivalent concentrations of 7–18 Bq m^{-3} and 0.3–0.6 Bq m^{-3} for radon and thoron, respectively.

(e) Norway

176. A study has recently been made in Norway of the radon concentrations in 129 dwellings in the area around Oslo [S24] (see Table 33). Radon daughter measurements were made in 25 dwellings to estimate the average equilibrium factor F which was found to be 0.5. Extensive measurements were made to study the effects of ventilation and changes in barometric pressure and the emanating power of some building materials.

177. The measurements were carried out during the winter of 1977–1978. The radon concentrations were found to be dependent on radium concentration and porosity of the building material and on radon exhalation from the ground, but were even more dependent on the ventilation and the atmospheric pressure. It appeared of special interest to evaluate the influence of ventilation on radon concentration during the winter season, when the ventilation rate is relatively low because of energy conservation efforts. The distribution of people living in various types of houses was assumed to be 20% of the people in wooden buildings, 5% in brick buildings and 75% in concrete buildings in the Oslo region. Over the whole country the corresponding values are 75%, 3% and 22%.

178. The distribution of radon concentrations (see Table 33) was approximately log-normal. On average, wooden houses had higher radon concentrations than brick houses, an observation explained by the fact that wooden houses in Norway are mostly one- or two-storey buildings for which radon emanation from the ground can be expected to be of greater significance.

179. The data on the distribution of radon concentrations in houses in the Oslo region, combined with the data on the distribution of houses of different building materials in Norway, lead to an average radon concentration of 52 Bq m^{-3} . If it is assumed that the equilibrium factor is 0.5, the equilibrium equivalent concentration of radon would be 26 Bq m^{-3} , corresponding to an annual effective dose equivalent of 1.6 mSv. However, this may be too high an estimate for the whole country, as gamma-ray measurements in Norway indicate that the natural gamma-radiation background is relatively high in the Oslo region, compared with the mean value for Norway.

(f) Poland

180. Data on radon and radon daughters activity concentrations in flats in Poland have been reported [G13]. The results are summarized in Table 34. The radon values were given as maximum values and the significance of the weighted average as a representative value for the flats is therefore uncertain. Other data on measurements in dwellings indicate higher average values. From measurements in Puławy, Czestochowa

and Warsaw in the Ursynów district an average equilibrium equivalent concentration of radon is estimated to be 17 Bq m^{-3} (equilibrium factor = 0.5) [B16, B17], corresponding to an annual effective dose equivalent of 1 mSv.

(g) Sweden

181. A number of measurements have been carried out in Sweden [S35]. Following measurements made by Hultqvist in 1956 [H17] in about 300 dwellings built before 1946, radon in houses was not considered a matter of serious concern in Sweden until the last few years. The main reason for the recent change in attitude is the continuing tendency, in the development of building standards, to lower ventilation rates and to improve the airtightness in order to conserve energy. The ventilation rate in houses built during the years up to 1950 was $0.8 \pm 0.5 \text{ h}^{-1}$ in apartments and $0.9 \pm 0.5 \text{ h}^{-1}$ in detached houses. During the 1950s the ventilation rate in new dwellings was $0.6 \pm 0.2 \text{ h}^{-1}$ and $0.8 \pm 0.2 \text{ h}^{-1}$; during the 1960s $0.5 \pm 0.2 \text{ h}^{-1}$ and $0.6 \pm 0.2 \text{ h}^{-1}$; and during the 1970s $0.3 \pm 0.15 \text{ h}^{-1}$ and $0.45 \pm 0.45 \text{ h}^{-1}$; in apartments and detached houses, respectively. This decrease in the ventilation rate causes an increased radon concentration in houses. Furthermore, more people live in apartment houses now than previously and modern detached houses contain more stone materials than older houses.

182. Up to 1975, a common building material was aerated concrete containing alum shale. This material contains more radium than other building materials and this has given rise to an increased exposure to gamma radiation and radon daughters. It is estimated that between 350 000 and 700 000 dwellings in Sweden contain aerated concrete based on alum shale. Of these, between 3000 and 20 000 are estimated to have radon daughter concentrations in indoor air requiring further investigation.

183. The radium concentration in this material varies as does the use of the material in the houses. From an investigation made in 1956 [H17] the average equilibrium equivalent radon concentration in these houses was estimated to be 58 Bq m^{-3} . The maximum value was about 5 times higher and the distribution of radon daughter concentrations was found to be more dependent on the ventilation than on the activity concentration in the building material.

184. However, some houses have been identified with an unusually large fraction of concrete based on alum shale in their building material. Measurements on radon and radon daughters have been made in 32 single-family houses where that is the case. The results are shown in Table 35 [S45]. The average of the radon daughter activity concentrations measured in all houses is 260 Bq m^{-3} , expressed as equilibrium equivalent concentration of radon. However, whether this average value is really representative of the true average is uncertain. There are reasons to believe that it may be an overestimate [S45].

185. In Sweden an unusual environmental situation has recently been described. Alum shale exists in many areas of central and southern Sweden and it has been used since the sixteenth century and up to the 1930s for the production of alum. It was also used for lime-

burning. There are therefore many tailings piles from this production, the number and location of which are not yet known. Some of them have been used as housing areas. Measurements have been carried out in one such area, outdoors and in houses built on the tailings [S34]. It has been estimated that 200 to 2000 houses may have been built in such areas [S29].

186. The radium concentration of the tailings was on average 2900 Bq kg^{-1} . The gamma radiation dose rate in air above reclaimed areas was about $0.4 \mu\text{Gy h}^{-1}$ and above unreclaimed areas about $1 \mu\text{Gy h}^{-1}$. In recreation areas the dose rates were $0.9\text{--}2.4 \mu\text{Gy h}^{-1}$. The radon concentration outdoors was $45\text{--}70 \text{ Bq m}^{-3}$ at the time of measurement and the radon concentration in drinking water was about 10 kBq m^{-3} . Measurements were carried out in 7 single-family houses. The average radon concentration was 860 Bq m^{-3} (range $430\text{--}2100 \text{ Bq m}^{-3}$) and the equilibrium equivalent concentration of radon was 500 Bq m^{-3} . The ventilation rate was low ($0.09\text{--}0.28 \text{ h}^{-1}$) and was in fact lower than the current Swedish standard (0.5 h^{-1}). If the ventilation rate were increased to 0.5 h^{-1} , the average radon concentration would be about 400 Bq m^{-3} , corresponding to an equilibrium equivalent concentration of radon of 200 Bq m^{-3} with an equilibrium factor of 0.5, provided that the increased ventilation rate does not cause a drop in pressure, which might effect the exhalation rate from the ground below.

187. In order to estimate the overall average radon daughter concentration in homes in Sweden, measurement of radon and radon daughters, gamma radiation, activity concentration in building material and of ventilation rates are being carried out in "typical" houses. The results are to be combined with other results on activity concentration in building materials, gamma radiation in houses, distribution of various ventilation systems and building materials, and variation of building practices with time. One part of the investigation includes 63 "typical" dwellings selected from seven types of houses built at the beginning of the 1970s in the town of Gävle in central Sweden. The results of the investigation are presented in Table 36 [E2].

188. The measurements on radon and radon daughters in houses in Sweden continue and new results indicate that the greatest problem is radon emanating from the ground. Houses have been found with more than $10\,000 \text{ Bq m}^{-3}$ of radon and it is estimated that about 75% of the total collective dose caused by inhalation of radon daughters depends on radon from the ground.

189. The estimated average radon concentrations for all homes in Sweden (dwellings existing in 1950, 1975 and estimates for 1985) are presented in Table 37 [S35]. Later recalculation has shown that the value for 1975 is probably underestimated by about 50% because of lower ventilation rates than assumed. Recent measurements in 600 houses representative for Sweden indicate an average value for radon daughters of 60 Bq m^{-3} in 1980. The national average of the equilibrium equivalent concentration is believed to lie within the extremes of 40 and 140 Bq m^{-3} . Therefore, the value for 1985 is probably also an underestimate. The values of the radon daughter concentrations are log-normally distributed. A concentration of 60 Bq m^{-3} corresponds to an annual effective dose equivalent of 3.7 mSv.

(h) United Kingdom

190. A wide-ranging survey in the United Kingdom has been reported [C6]. The measurements were carried out in 87 dwellings in England and Scotland varying in age from one to 300 years. They were in the main single-family houses made of clay brick, many with concrete floors at ground level. Measurements were made of the activity concentration of ^{218}Po and from these measurements, together with determination of ventilation rates, the radon exhalation coefficients (see paragraph 34) were calculated. Because the exhalation coefficient is practically independent of the ventilation rate within the range of variation occurring in houses, it was possible to calculate the radon daughter concentrations corresponding to an average ventilation rate in the houses, assumed to be 1 h^{-1} .

191. The median value for radon exhalation coefficient was found to be $11.8 \text{ Bq m}^{-3} \text{ h}^{-1}$ and the arithmetic mean was $22.2 \text{ Bq m}^{-3} \text{ h}^{-1}$, with a range from 0.44 to $204 \text{ Bq m}^{-3} \text{ h}^{-1}$. By excluding the extreme maximum value from a rather uncommon type of dwelling the arithmetic mean was reduced to $20 \text{ Bq m}^{-3} \text{ h}^{-1}$. (The exhalation rate expressed in units of $\text{Bq m}^{-2} \text{ h}^{-1}$ can be arrived at by dividing by 2 m^{-1} , which is the appropriate value of the expression $\frac{S}{V}$, where S is the surface of walls, floors, etc., and V is the volume of rooms.)

192. Assuming an average ventilation rate of 1 h^{-1} and paying due consideration to the radon concentration in the outside air—assumed to be 3 Bq m^{-3} —the average equilibrium equivalent radon concentration in the houses was found to be 13 Bq m^{-3} . That corresponds to an annual effective dose equivalent rate of 0.8 mSv . The cumulative frequency distribution of dose from radon decay products is shown in Figure XXXII.

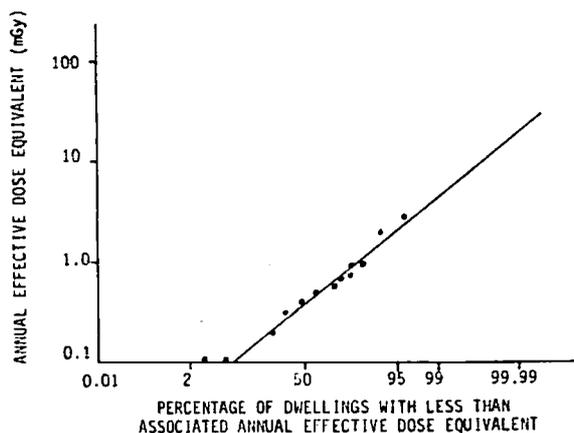


Figure XXXII. Cumulative frequency distribution of annual effective dose equivalent from radon decay products in some dwellings in the United Kingdom

193. Phosphogypsum can replace ordinary material in buildings. The radium concentration of phosphogypsum varies between $20\text{--}1500 \text{ Bq kg}^{-1}$ (see Annex C). In a prospective study, the radon concentration in houses with phosphogypsum (radium concentration 900 Bq kg^{-1}) has been estimated to be about 7 Bq m^{-3} [O3, W20]. The ventilation rate was assumed to be 1 h^{-1} .

(i) United States

194. The distribution of radon and radon daughter activity concentrations has been investigated over a two-year period in 21 New Jersey and New York residences [G1]. Most of the buildings were single-family, of one or two stores, and of wood frame or brick construction. The geometric mean of the equilibrium equivalent radon concentration was 15.2 Bq m^{-3} , and the arithmetic mean 16.3 Bq m^{-3} (range 2.6 to 107 Bq m^{-3}) corresponding to an annual effective dose equivalent of 1 mSv .

195. Radon daughter concentrations in houses on phosphate-related land in Florida have been reported [U4, U8]. There are 1000–3000 houses built on phosphate lands and a classification of buildings and their equilibrium equivalent radon concentrations based on radon daughter measurements in Polk County are shown in Table 38 [U8]. In Colorado some waste products from the uranium industry have been used as filling material under a number of houses causing an enhanced radon concentration indoors. The radium concentration in this material was 4400 Bq kg^{-1} on average. Table 39 shows the distribution of equilibrium equivalent radon concentration in some of these houses [C11]. The average value estimated from the table is about 200 Bq m^{-3} ; the weighted average for a population of 15 000 is about 37 Bq m^{-3} , corresponding to an annual effective dose equivalent of 2.3 mSv .

196. High radon concentrations have also been found in houses having unpaved crawl spaces. In a total of 22 houses investigated in the Chicago area, the radon concentration was more than 185 Bq m^{-3} in nine houses and six of these houses had more than 370 Bq m^{-3} . The highest values were about 1000 Bq m^{-3} . The radon emanated from the unpaved crawl space under the house and the exhalation rate was about $0.3 \text{ Bq m}^{-2} \text{ s}^{-1}$. The radium concentration in the soil was normal (about 40 Bq kg^{-1}) [R12].

(j) Yugoslavia

197. In the neighbourhood of uranium ore deposits in Yugoslavia, radon concentrations have been measured in houses, some of which were built of stone from the uranium area [K20]. The concentrations were in the range of $30\text{--}100 \text{ Bq m}^{-3}$. All measurements were made in the daytime and because the radon concentrations were found to increase during the night the reported values underestimate the average by approximately a factor of 2. The radon concentration outdoors varied between 4 and 8 Bq m^{-3} .

(k) Other countries or areas

198. Other measurements were reported in Annex B of the 1977 report [U6]. There have also been estimates of the radon daughter exposures in houses, on the basis of the radium concentration in building materials [N2]. Using a model developed by Krisiuk et al. [K13] and applying "typical" values of radium concentration in building material (50 Bq kg^{-1}), an emanating power of $0.01\text{--}0.04$, and a ventilation rate (1 h^{-1}), estimated typical values were obtained for the increases in exposure to radon daughters above outdoor levels. The results were given in WLM. These results are given in this Annex recalculated as average values of equilibrium equivalent concentration of radon. The equilibrium equivalent concentration for outdoor air is added, namely 1.8 Bq m^{-3} . These results, together with

those given in Annex B of the 1977 report [U6], are included in Table 40.

(1) Summary

199. The distribution of the radon daughter concentrations in houses is generally log-normal. With the exception of Sweden, 90% of buildings have concentrations less than about 50 Bq m^{-3} . A few per cent may have values greater than 100 Bq m^{-3} . Sweden seems to be exceptional with more than 30% of buildings with indoor radon daughter concentrations above 100 Bq m^{-3} . The main source of radon is the soil and building materials (aerated concrete) containing alum shale. Another reason is the reduced ventilation rate in all houses over the last few years, as a result of energy conservation programmes.

200. Besides the general distribution of radon daughter concentration, there are in many countries exceptionally high levels because of high radium concentration in the ground or building materials, sometimes in combination with poor ventilation. Extreme values between 1000 and 10 000 Bq m^{-3} occur. The relative number of houses affected by extreme values ($> 1000 \text{ Bq m}^{-3}$), such as some in Grand Junction and Florida in the United States, in Sweden and the United Kingdom, may be between 0.01 to 0.1%. The corresponding annual effective dose equivalent is 60 mSv or more.

201. However, the main contribution to the collective effective dose equivalent may not result from the small number of houses with elevated radon concentrations. If it is assumed that the radon daughter activity concentration is less than 50 Bq m^{-3} for 90% of the people, $50\text{--}100 \text{ Bq m}^{-3}$ for 9%, $100\text{--}1000 \text{ Bq m}^{-3}$ for 0.9% and $1000\text{--}10\ 000 \text{ Bq m}^{-3}$ for 0.09% and the middle value in each range is used, the collective effective dose equivalent caused by exposure to radon daughter concentrations of less than 100 Bq m^{-3} correspond to about 75% of the total collective effective dose equivalent. For Sweden this estimate is not valid, as a larger fraction of the population is exposed to high radon daughter concentrations.

202. The average values of the equilibrium equivalent concentrations of radon in dwellings in different countries are summarized in Table 40, where the corresponding effective dose equivalents are calculated using the dosimetric coefficients given in Table 22. The values should be considered with some caution, as seasonal variations are, for instance, not taken into account.

203. It may be seen in Table 40 that with the exception of Sweden the mean values of the indoor equilibrium equivalent ^{222}Rn concentration in different countries cover the range from 5 to 25 Bq m^{-3} . Taking into account the dose conversion coefficients listed in Table 22, this corresponds to an annual effective dose equivalent ranging from 0.3 to 1.5 mSv. For the total population in the temperate regions of the world, 15 Bq m^{-3} seems to be an appropriate mean value for the indoor concentration, which is about 8 times higher than the mean activity concentration in outdoor air (1.8 Bq m^{-3}). This yields a mean annual effective dose equivalent of about 0.92 mSv from indoor exposure and of about 0.06 mSv from outdoor exposure, giving a total of about 1 mSv from inhaled ^{222}Rn daughters. This value refers to temperate regions of the world. For equatorial regions so far no measurements are

available. Having regard to the different domestic conditions, the indoor concentrations of radon daughters in those regions might be considerably lower than in temperate regions. For large population groups this level will be comparable with the normal outdoor level, leading to an annual effective dose equivalent of about 0.2 mSv. Taking into account that about two-thirds of the total world population is living in temperate regions, a global mean annual effective dose equivalent of about 0.8 mSv—averaged over all age groups—from inhaled ^{222}Rn daughters would be expected (See Table 12 of Annex B).

204. By comparison with ^{222}Rn daughters, very few measurements of ^{220}Rn daughters have been reported [C6, S21, W14]. However, the simultaneous measurements of ^{222}Rn and ^{220}Rn daughters in houses in the Federal Republic of Germany (see Table 32) and in the United Kingdom (see Table 41) seem to indicate a ratio of about 20 between equilibrium equivalent ^{222}Rn concentration and equilibrium equivalent ^{220}Rn concentration. By applying this factor, it can be concluded that the mean value of the indoor equilibrium equivalent ^{220}Rn concentration in different countries should be in the range of 0.2–1.2 Bq m^{-3} . For the total population in temperate regions of the world a mean value of about 0.7 Bq m^{-3} could be expected. Taking into account the dosimetric coefficients for ^{220}Rn daughters given in Table 22, this corresponds to a mean annual effective dose equivalent of about 0.2 mSv from inhaled ^{220}Rn daughters. This means that the contribution from ^{220}Rn daughters to the effective dose equivalent is on the average about one-fifth of that from ^{222}Rn daughters. For reasons mentioned in paragraph 203, a global mean value of about 0.17 mSv per year from inhaled ^{220}Rn daughters should be expected (see Table 16 of Annex B).

D. OCCUPATIONAL EXPOSURES TO RADON AND THORON DAUGHTERS

1. Uranium mines

205. In Annex B of the 1977 report [U6], some data were reported about radon and radon daughter concentrations in uranium mines and it was noted that the improvement in working conditions noticed during the previous years was continuing. This is still the case, although the technical difficulties involved in making further improvements increase as the concentrations are decreased. Radon daughter concentrations and exposures in mines in some countries are shown in Table 42. The value for uranium mines in France in 1979 is calculated by using an equilibrium factor of 0.17. By using an effective dose equivalent per unit of potential alpha-energy exposure of 8.4 mSv WLM^{-1} (see Table 19), the annual effective dose equivalent for uranium miners is estimated to be 34 mSv in the United States in 1977, 12 mSv in France in 1979 and 6.2 mSv in Canada in 1979. The average annual effective dose equivalent, including the data from India given in paragraph 208, is 15 mSv.

206. The activity concentration of thoron daughters is not always insignificant. In measurements of radon daughters and other radiation variables in the presence of ^{220}Rn at Rio Algom mine, Canada, the representative average value for radon daughters was 0.084 WL and for thoron daughters 0.12 WL [B18]. Similar measurements have also been made in the Agnew Lake mine, Canada, and the corresponding values were 0.1 WL for

radon daughters and 0.09 WL for thoron daughters [B19]. Because the thoron daughter concentration is comparable to that of radon daughters, special consideration should also be given to the measuring technique [C18].

207. Radon daughter activity concentration in open-pit mines and its variation with meteorological variables have been studied at Nabarlek uranium mine, Australia [L13], which is a high grade (2%) uranium ore mine. The radon exhalation rate varied widely but the quotient of exhalation rate to ore grade was in general fairly stable, about $80 \text{ Bq m}^{-2} \text{ s}^{-1}$ per percentage of U_3O_8 . The concentration of radon and radon daughters in air was strongly influenced by air movements and atmospheric stability and the variations could be up to an order of magnitude. Normally the radon concentrations were lower during daylight than during the night. However, during the hours just before dawn the air frequently becomes still and the radon and radon daughter concentrations rise to a maximum. The estimated radon daughter exposure during about half a year was 0.065 WLM, corresponding to an effective dose equivalent of 0.5 mSv.

208. Radon daughter measurements have been made in Jaduguda underground mines in India on a regular basis for many years. The radon daughter exposure has been estimated for different kinds of operations. The highest exposures occur most frequently during drilling. The results for India are presented in Table 43 [R13, K19, A12]. The average effective dose equivalents for the different categories of mine workers ranged from 14 to 22 mSv in 1979.

209. As a curiosity, the radon concentrations in the well-known old mines in Schneeberg and Jachymov may be mentioned. The measurements were performed at the beginning of this century and published in the 1920s [B21, L14]. The concentrations were given in "Mache" units, where 1 Mache = 13.3 Bq. The radon concentrations in various parts of Schneeberg mine ranged from 20 to 600 kBq m^{-3} and the average was about 100 kBq m^{-3} . In Jachymov mine the corresponding values were 10–300 kBq m^{-3} and 100 kBq m^{-3} . These average values correspond to about 30 WL.

2. Non-uranium mines

210. Since radon problems in many non-uranium mines were highlighted during the 1960s and the 1970s, the radon daughter concentration in these mines has decreased continuously, as can be seen in Table 44. The corresponding annual effective dose equivalents are for Finland (1977) 3.2 mSv, Norway (1980) 3.8 mSv, South Africa (1973) 14 mSv, Sweden (1980) 5.9 mSv, and for the United Kingdom (1981) about 1.0 mSv for most of the coal miners and 22 mSv for miners in other mines. In a research report on the health effects of radon exposure in non-uranium mines in Bavaria, Federal Republic of Germany, the radon concentrations in nine mines are reported [F7] for the years 1971–1973. The values reported correspond to potential alpha-energy exposures between 0.1 and 4 WLM, if an equilibrium factor of 0.7 is used.

211. Thoron daughter measurements have only been carried out in a few mines. In two non-uranium mines in Norway the average equilibrium equivalent concentrations of thoron were about 3 Bq m^{-3} and 20 Bq m^{-3} [B4]. In the United Kingdom the mean ^{212}Pb concentra-

tions in seven metalliferous mines were 40, 20, 3, 1, 1, 1 and 0.4 Bq m^{-3} [D6]. Accordingly, the equilibrium equivalent concentration of thoron in mines seems to be in the range of about 4 to 40 Bq m^{-3} , corresponding to an annual effective dose equivalent of 0.5 to 5 mSv.

3. Other occupational exposures

212. Radon exhalation from ordinary rocks and soils and radon-rich water can cause high radon and radon daughter concentrations in underground spaces such as tunnels, hydroelectric power stations, caves, public baths and thermal spas. Often an occupational exposure occurs only for a fraction of the working time and this fraction is sometimes difficult to estimate. Table 45 shows concentrations of, and exposures to, radon daughters. In some cases the exposures are rough estimates made by assuming a 2000 h exposure per year; in other instances the real working time has been considered. The average annual exposures to radon daughters in spas have been studied over a period of between 15 and 32 years in the island of Ischia in Italy [B9]. The corresponding annual effective dose equivalents range from 1 to 120 mSv.

213. The levels and distribution of radon concentration have been measured in two calcite caves in Japan, Akiyoshi Cave and Kagekiyo Cave [M23]. The radon concentration increases by one order of magnitude between a point near the entrance and at points further away. The average value given in Table 45 (0.8 WL) refers to a point 400 m from the entrance. Great seasonal variations occur with substantial increases in the summer; a factor of 500 has been reported. The value in Table 45 refers to March 1978. The average radon exhalation rate was estimated to be about $7 \cdot 10^{-3} \text{ Bq m}^{-2} \text{ s}^{-1}$ and the radium concentration in the cave wall was reported to be lower than that in the soil outside the cave.

214. In factories where work with thorium is carried out (thorium extraction from ore, manufacture of gas mantles, production of Mg/Th alloys) there is occupational exposure to thoron and thoron daughters. In some thorium factories in the United Kingdom, measurements were performed of thoron and thoron daughters [D6]. In one gas mantle factory extensive observations were made of the ^{220}Rn : ^{212}Pb : ^{212}Bi ratios, absolute values and variations of the concentrations. Since such measurements are rare, they are shown fully in Table 46. The "inferred values" of thoron are those estimated on the basis of the ^{212}Pb and ^{212}Bi concentrations. There is little regularity in the activity concentration ratios, probably on account of sharp thoron gradients in the rooms. The thoron concentrations given in Table 46 are rather typical of some of the other factories examined. Most values are in the range of 1–10 kBq m^{-3} . Other factories had lower values, around 0.4–4 kBq m^{-3} . These values are rather similar to others reported (2–20 kBq m^{-3}) from two thorium plants in India [M16]. The average value of all ^{212}Pb measurements in the United Kingdom [D6] was 170 Bq m^{-3} , corresponding to an annual effective dose equivalent of about 20 mSv. Some typical variations of the potential alpha-energy concentration of thoron daughters in a thorium factory in India are shown in Table 47 [D10]. The average value is 0.15 WL and the corresponding exposure is less than 1.8 WLM, corresponding to an annual effective dose equivalent of about 5 mSv.

E. DELIBERATE EXPOSURES TO RADON

215. For several hundred years radon has been exploited in "balneology"—the use of spas or baths for the alleviation of illness. There are balneological facilities using radon in Badgastein, Austria, in Bulgaria, in Poland and in the USSR.

216. There is a large amount of information about the levels of radon in Badgastein, much of which has been summarized by Uzunov et al. [U10]; they have estimated the doses received by the general population of the town, by the employees of the spa and by the individuals deliberately exposed to the radon in the course of balneological procedures.

217. Thermal springs in the centre of the town supply radon-rich hot water at a concentration of about 10^6 Bq m^{-3} . Annually about $2 \cdot 10^{12}$ Bq of radon diffuse from the water into the air, resulting in a mean indoor concentration of about 300 Bq m^{-3} in buildings situated in the central part of the town, and up to 10^5 Bq m^{-3} in the spa facilities [U10].

218. The annual exposures of employees of the spa, who inhale air with radon concentrations of up to 10^5 Bq m^{-3} , are estimated to be between 1 and 40 WLM [U10], corresponding to an effective dose equivalent of 8–300 mSv.

219. Individuals undergoing deliberate exposure include children recovering from poliomyelitis and other neurological diseases, who may spend up to several years in a special children's sanatorium. Their annual exposure is estimated to be about 0.7 WLM [U10] corresponding to an effective dose equivalent of about 4 mSv. Adults who breathe the radon-rich air in the inhalation facilities of the spa are estimated to receive 0.2 WLM from the standard two-week course, which involves a total exposure time of about four hours.

220. Uzunov et al. [U10] conclude their paper with the following statement: "The highly positive effects of balneological treatment for man as claimed by balneologists are not questioned by us. However, whether radon represents a necessary and justifiable component seems to be doubtful. Considering the world-wide attempts to reduce deliberate irradiation of man by ionizing radiation it seems an apparent anachronism to expose significant population groups, to a large extent, even uncontrolled, to one of the strongest known carcinogens, i.e. the atmospheric alpha emitting radon decay products."

VI. ENERGY CONSERVATION AND LEVELS OF RADON IN AIR

221. In recent years many countries have established significant programmes to conserve energy. This chapter will consider the consequences of energy conservation in terms of effective dose equivalents caused by inhalation of radon and radon daughters in the air.

222. Energy conservation in industry means a possible increase in the effectiveness of the use of machines and rooms and a decrease in energy consumption by maintenance and service systems. The ventilation system may be switched off or used at a reduced level, in compliance with regulations

prescribed by the authorities. In general, a reduction of the ventilation rate increases the radon and radon daughter exposure, unless special countermeasures are taken. For example, reduced ventilation during nights and weekends requires consideration of the minimum time for ventilation before work starts. In mines particularly, the radon and radon daughter concentrations may increase by several orders of magnitude when ventilation is reduced and several hours of maximum ventilation may be necessary before work starts [S16]. It can be inferred from Table 19 that an increase of 1 Bq m^{-3} in the average equilibrium equivalent radon concentration in a mine leads to an increase in the annual effective dose equivalent of about 30 μ Sv.

223. In houses, energy conservation may involve reduction of ventilation by sealing of windows and doors or by reduction of exhaust ventilation. These measures enhance the radon concentrations approximately in inverse proportion to the ventilation rates (for ventilation rates $\lambda_v > 0.1$ h^{-1}). Other procedures which will influence the radon concentrations are recirculation of the air to reduce the effective ventilation, and the use of "heat-conserving beds", i.e., the circulation of air through pieces of rock under the house, which will enhance the radon concentration because of exhalation from the rock [N3, S47]. Similarly, a reduction in the admission of outside air by a cooling system in a house will reduce the ventilation rate.

224. The principal methods of calculating the increased radon and radon daughter concentrations as a result of decreased ventilation rates in an energy conservation programme will be given for houses. The radiological impact is expressed as the excess collective effective dose equivalent (man Sv). Decisions on energy conservation may relate to existing houses or new houses and practical energy conservation programmes may be different in these two cases. However, the methods of calculation of doses with and without a defined energy conservation programme are in principle the same in both cases.

225. To estimate the radiological impact, the difference has to be calculated between the collective doses before and after the programme has been realized. As the ventilation decreases from λ_{v1} to λ_{v2} (h^{-1}) the radon activity concentration increases from $\chi_{a,Rn,1}$ to $\chi_{a,Rn,2}$ (Bq m^{-3}) according to the formulae

$$\chi_{a, Rn, 1} = \frac{A/V}{\lambda_{v1}} \quad (33)$$

(the small correction required by a normal radon concentration outdoors has been neglected)

$$\chi_{a, Rn, 2} = \frac{A/V}{\lambda_{v2}} \quad (34)$$

$$\Delta\chi_{a, Rn} = \frac{A}{V} \frac{\lambda_{v1} - \lambda_{v2}}{\lambda_{v1} \lambda_{v2}} \quad (35)$$

where A/V is the source term (Bq $m^{-3} h^{-1}$) including radon emanation (Bq h^{-1}) from building materials, the ground under the house, and releases from water and gas as seen in equation (21).

226. The number of people exposed are grouped according to the type of house (source terms and venti-

lation) and, after measurements or calculations of average radon and radon daughter concentration in these types of houses, the collective dose can be estimated. As can be seen from equation (35), as long as the source term remains constant and does not change as a consequence of an energy conservation programme, the change in radon concentration depends only on the ventilation rates. The radon daughter concentration (and hence the equilibrium factor F) will increase because of the decreased ventilation rate. Table 5 shows that the factor F will increase as λ_v decreases, if the deposition effect is not taken into account. An estimation which only takes into account the relative increase in the radon concentration will therefore underestimate the impact of reduced ventilation, as will be shown below.

227. The dose caused by inhalation of radon daughters in air with radon concentration $\chi_{a,Rn}$ is $J_c F \chi_{a,Rn}$ where J_c is the dose conversion coefficient and F the equilibrium factor. If the ventilation is changed from λ_{v1} to λ_{v2} the increase in the dose will be

$$\begin{aligned} (\Delta D)_1 &= D_2 - D_1 = J_c \frac{A}{V} \left[\frac{F_2}{\lambda_{v2}} - \frac{F_1}{\lambda_{v1}} \right] = \\ &= J_c \frac{A}{V} F_1 \frac{\frac{F_2}{F_1} \lambda_{v1} - \lambda_{v2}}{\lambda_{v1} \lambda_{v2}} \end{aligned} \quad (36)$$

If it is assumed that the equilibrium factor is unchanged and has the value F , the estimated increase in the dose will be

$$\begin{aligned} (\Delta D)_2 &= D_2 - D_1 = J_c \frac{A}{V} F_3 \left[\frac{1}{\lambda_{v2}} - \frac{1}{\lambda_{v1}} \right] = \\ &= J_c \frac{A}{V} F \frac{\lambda_{v1} - \lambda_{v2}}{\lambda_{v1} \lambda_{v2}} \end{aligned} \quad (37)$$

The ratio $(\Delta D)_1/(\Delta D)_2$ is a measure of the degree of underestimation. For values $0.5 < \lambda_v < 1 \text{ h}^{-1}$ and a value $F = 0.5$ (as recommended in this Annex, paragraph 136) the underestimation will be less than about a factor of 2.5. The formulae and conclusions above do not include houses with gas heating in which case, as mentioned earlier, the radon release rate will decrease because of the reduced consumption of gas. However, as can be seen in Table 13, the relative contribution of radon from gas may be of little significance.

228. As an example, the radiological consequences of a proposed energy conservation programme in Sweden is given [S36]. In Sweden the total electrical energy consumption (transformation and transfer losses excluded) was about 45 GW a in 1977. About 10 GW a was used in dwellings and of that about 7 GW a was used for heating. Of that, 25% was lost by ventilation. It has been postulated that significant amounts of energy could be conserved by reducing the ventilation and improving the ventilation systems. A decrease in the ventilation rate by 0.2–0.3 h^{-1} has been discussed.

229. By measuring ventilation and radon and radon daughters in different types of houses and building materials, it has been possible to estimate the average radon concentration and collective effective dose equivalents and their change with a change in the ventilation rate. The calculations have only included houses with natural draught ventilation systems and those with a

ventilation rate greater than 0.5 h^{-1} . One alternative in the energy conservation programme was to reduce ventilation to a rate not lower than 0.5 h^{-1} . The results of comparing the amount of energy conserved and the resultant increase in the collective doses, are presented in Table 48 (after recalculation of the values in terms of effective dose equivalents and using the dosimetric coefficients given in Table 22). The equilibrium factor F is assumed to be 0.5 and unchanged. The values in Table 48 can be compared with the total annual collective effective dose equivalent from houses in 1975 of about 18 000 man Sv ($8.2 \cdot 10^6$ persons multiplied by 2.2 mSv a^{-1}), assuming a radon daughter concentration of 36 Bq m^{-3} (see Table 37). Assuming that the weighted average figure of 5.6 man Sv (MW a^{-1}) applies generally, the relative increase in individual annual effective dose equivalent per MW a of electrical energy saved is about 0.03%. The effects of the radon activity concentration in air in houses caused by an energy conservation programme is much dependent on local circumstances. The example given above from Sweden is relevant only for Swedish conditions because of existing high normal radon concentrations and low ventilation rates. However, energy conserved by decreasing the ventilation will always increase the radon and radon daughter concentration.

VII. SUMMARY

230. Radon and thoron are naturally-occurring radioactive gases, which are products of the uranium and thorium decay series, respectively. Uranium and thorium occur widely in the environment, in rock, soil, air, water, building materials, man, etc. Some of the radon and thoron diffuses from the material in which it is formed, is dispersed in ground water and in air. The total equilibrium amount of radon in air is of the order of 10^{18} Bq. The corresponding value of thoron is several orders of magnitude less. Of the total global inventory of radon, radium in soil contributes two orders of magnitude more than any other source.

231. The decay of radon and thoron and the subsequent decay of their daughters terminates with the formation of stable lead. The radon and thoron daughters occur in the same media as their precursors. It takes some time, of the order of one hour, for the daughter products to reach equilibrium. Because of this, and because of deposition on surfaces, there is often less than equilibrium amounts of the daughter products in air and water. The equilibrium factor is a measure of this deficit in air. Radon and thoron daughters in air are predominantly attached to aerosols. A minor part, normally less than 10%, occur as unattached atoms or ions. The relative distribution of attached and unattached daughters in air and the equilibrium factor depend on many variables, such as the decay constant, the concentration and size distribution of aerosols and the ventilation rates. Increased ventilation decreases the concentration of radon and thoron daughters in air.

232. The distribution of radon, thoron and their decay products in outdoor air depends on the vertical temperature gradient, the direction and strength of the wind and the air turbulence. Owing to its short half-life, thoron occurs only within a few tens of metres above ground, while radon occurs up to an altitude of several kilometres. The concentration at ground level depends on meteorological conditions and geographical location. Normally a minimum concentration occurs in the spring and summer and a maximum in the autumn

and early winter. Mean annual values of radon concentration in outdoor air vary between 0.1 and 10 Bq m⁻³. The higher concentrations are found in air above continental areas and the lower concentrations in air over arctic areas and above the sea. An average value of 3 Bq m⁻³ over land and an equilibrium factor of 0.6 or less for radon daughters outdoors is probably a reasonable estimate. The concentration of ²¹²Pb is normally about one order of magnitude less than that of radon.

233. In closed spaces, e.g., a mine or a house, the concentrations in air of radon, thoron and their decay products are higher than outdoors. In houses the radon daughter levels may be enhanced by radon from radium-rich building materials, landfill, soil and bedrock under the house, radon-rich water and by poor ventilation. The radon activity concentration in water varies from practically zero to very high values of the order of 100 MBq m⁻³. The radon levels indoors may be enhanced by any of these sources and reach values as high as 10 000 Bq m⁻³. Normal values of radon daughter activity concentration indoors are of the order of 20 Bq m⁻³. Annual occupational exposure to radon daughters in mines and other working places underground is generally less than 4 WLM corresponding to an equilibrium equivalent concentration of radon of about 1000 Bq m⁻³. In well-ventilated mines the equilibrium factor is low (< 0.5). In unventilated parts of mines radon concentrations as high as 1 MBq m⁻³ or even more may occur.

234. Inhalation of radon and thoron daughters leads to deposition in the human respiratory tract and consequent irradiation. The deposition depends on various factors, such as the size distribution of the aerosols to which the daughter products of radon and thoron are attached, and the fraction of unattached daughters. On average, the dose to the bronchial basal cell layer in the lung is about 5 to 8 times higher than the dose to the pulmonary region. The effective dose equivalent for radon and thoron daughter exposures may be calculated using weighting factors for the regional distribution of lung dose and the mean lung dose. The Committee's present estimate of mean annual effective dose equivalent is 1 mSv and arises mostly from radon in houses. There is little experimental information to provide a good estimate of the annual effective dose equivalent from thoron daughters; a value of 0.2 mSv is tentatively proposed. The radon and thoron daughter concentrations in air, and resulting doses given in this Annex refer mainly to countries in temperate parts of the world. Taking into account that about two-thirds of the total world population is living in temperate regions, a global mean value of about 0.8 mSv per year—averaged over all age groups—from inhaled ²²²Rn daughters and of about 0.17 mSv per year from inhaled ²²⁰Rn daughters would be expected. Occupational exposure to radon daughters in uranium mines causes average annual effective dose equivalents of about 15 mSv, and similar exposures have been encountered in some non-uranium mines.

Table 1
Radioactive decay properties of ^{226}Ra and its daughters
 [E5, L12]

Radionuclide	Historical name	Half-life	Major radiation energies and intensities					
			α		β		γ	
			MeV	%	MeV	%	MeV	%
$^{226}_{88}\text{Ra}$	Radium	$1.6 \cdot 10^3$ a	4.60	6			0.186	3.3
\downarrow			4.78	94				
$^{222}_{86}\text{Rn}$	Emanation Radon (Rn)	3.823 d	5.49	100				
\downarrow								
$^{218}_{84}\text{Po}$	Radium A	3.05 min	6.00	~100				
\downarrow								
99.98% \downarrow $^{214}_{82}\text{Pb}$	Radium B	26.8 min			0.67	48	0.295	19
					0.73	42	0.352	37
					1.02	6		
0.02% \downarrow $^{218}_{85}\text{At}$	Astatine	~ 2 s	6.65	6	?	~ 0.1		
			6.69	90				
			6.76	3.6				
\downarrow								
$^{214}_{83}\text{Bi}$	Radium C	19.7 min	5.45	0.012	1.0	23	0.609	46
			5.51	0.008	1.51	40	1.12	15
					3.26	19	1.764	16
99.98% \downarrow $^{214}_{84}\text{Po}$	Radium C'	164 μs	7.69	100				
0.02% \downarrow $^{210}_{81}\text{Tl}$	Radium C''	1.3 min			1.3	25	0.296	80
					1.9	56	0.795	100
					2.3	19	1.31	21
\downarrow								
$^{210}_{82}\text{Pb}$	Radium D	22.3 a			0.015	81	0.047	4
					0.061	19		
\downarrow								
$^{210}_{83}\text{Bi}$	Radium E	5.01 d			1.161	~100		
\downarrow								
~100% \downarrow $^{210}_{84}\text{Po}$	Radium F	138.4 d	5.305	100				
0.0001% \downarrow $^{206}_{81}\text{Tl}$	Radium E''	4.2 min			1.53	100		
\downarrow								
$^{206}_{82}\text{Pb}$	Radium G	Stable						

Radioactive decay properties of ^{228}Th and its daughters
 [E5, L12]

Radionuclide	Historical name	Half-life	Major radiation energies and intensities					
			α		β		γ	
			MeV	%	MeV	%	MeV	%
$^{228}_{90}\text{Th}$	Radiothorium	1.913 a	5.34	27			0.084	1.2
			5.43	73			0.216	0.3
\downarrow								
$^{224}_{88}\text{Ra}$	Thorium X	3.66 d	5.45	6			0.241	3.9
			5.68	94				
\downarrow								
$^{220}_{86}\text{Rn}$	Emanation Thoron (Tn)	55 s	6.29	100			0.55	0.1
\downarrow								
$^{216}_{84}\text{Po}$	Thorium A	0.15 s	6.78	100				
\downarrow								
$^{212}_{82}\text{Pb}$	Thorium B	10.64 h			0.331	83	0.239	43
					0.569	12	0.300	3.2
\downarrow								

Table 1, continued

Radionuclide	Historical name	Half-life	Major radiation energies and intensities					
			α		β		γ	
			MeV	%	MeV	%	MeV	%
$^{212}_{83}\text{Bi}$ ↓ 64% $^{212}_{84}\text{Po}$ ↓ 36% $^{208}_{81}\text{Tl}$ ↓ $^{208}_{82}\text{Pb}$	Thorium C	60.6 min	6.05 6.09	25 10	1.55 2.26	5 55	0.040 0.727 1.620	1.1 11.8 2.8
	Thorium C'	304 ns	8.78	100				
	Thorium C''	3.05 min			1.28 1.52 1.80	23 22 51	0.511 0.583 0.860 2.614	23 86 12 100
	Thorium D	Stable						

Table 2

Properties of radon
{J7, H4, W3}

Property	Value
Volume of 1 Bq of ^{222}Rn at NTP	$1.6 \cdot 10^{-20} \text{ m}^3$
Boiling point	-61.8° C
Melting point	-71° C
Vapour pressure at	
- 144° C	0.13 kPa
- 126.3° C	1.3 kPa
- 111.3° C	5.3 kPa
- 99° C	13 kPa
- 71° C	53 kPa
- 61.8° C	100 kPa
Density at NTP	9.96 kg m^{-3}
Coefficients of solubility ^{a/} at atmospheric pressure	
in water at	
0 $^\circ \text{ C}$	0.507
10 $^\circ \text{ C}$	0.340
20 $^\circ \text{ C}$	0.250
30 $^\circ \text{ C}$	0.195
37 $^\circ \text{ C}$	0.167
50 $^\circ \text{ C}$	0.138
75 $^\circ \text{ C}$	0.114
100 $^\circ \text{ C}$	0.106
	at 18 $^\circ \text{ C}$ at 0 $^\circ \text{ C}$
in glycerine	0.21 -
aniline	3.80 4.45
absolute alcohol	6.17 8.28
acetone	6.30 7.99
ethyl acetate	7.35 9.41
petroleum (liquid paraffin)	9.20 12.6
xylene	12.75 -
benzene	12.82 -
toluene	13.24 18.4
chloroform	15.08 20.5
ether	15.08 20.09
hexane	16.56 23.4
carbon bisulphide	23.14 33.4
olive oil	29.00 -

^{a/} The solubility coefficient or partition coefficient in water is defined as the ratio x_w/x_a , where x_w and x_a are the radon concentrations in water and air, respectively.

T a b l e 3

Calculated potential alpha energy of radon and thoron
and their short-lived decay products

Radio-nuclide	Potential alpha energy per				Potential alpha energy concentration (C _{pot}) per
	atom		unit of activity		unit of activity concentration
	E _{pot,at}		E _{pot,at} /λ _j		
	MeV	10 ⁻¹² J	MeV Bq ⁻¹	10 ⁻¹⁰ J Bq ⁻¹	10 ⁻⁶ WL (Bq m ⁻³) ⁻¹
²²² Rn	19.2	3.07	9150000	14700	
²¹⁸ Po	13.7	2.19	3620	5.79	27.8
²¹⁴ Pb	7.69	1.23	17800	28.6	137
²¹⁴ Bi	7.69	1.23	13100	21.0	101
²¹⁴ Po	7.69	1.23	0.002	0.000003	0.000016
Total (Rounded) a/			34500	55.4	266
²²⁰ Rn	20.9	3.34	1660	2.65	
²¹⁶ Po	14.6	2.34	3.32	0.00532	0.0256
²¹² Pb	7.8	1.25	431000	691	3320
²¹² Bi	7.8	1.25	40900	65.6	315
²¹² Po	8.78	1.41	0.00000305	0.000000062	0.00000003
Total (Rounded) a/			472000	757	3640

a/ The total is the sum of the potential alpha energies of the daughters only.

T a b l e 4

Calculated equilibrium equivalent concentration of radon and thoron
as a function of the potential alpha energy concentration C_{pot}

Radon x _{eq,Rn} (Bq m ⁻³)		Thoron x _{eq,Tn} (Bq m ⁻³)	
2.90 10 ⁻⁵ (Bq MeV ⁻¹) C _{pot} (MeV m ⁻³)		2.12 10 ⁻⁶ (Bq MeV ⁻¹) C _{pot} (MeV m ⁻³)	
1.81 10 ⁸ (Bq J ⁻¹) C _{pot} (J m ⁻³)		1.32 10 ⁷ (Bq J ⁻¹) C _{pot} (J m ⁻³)	
3700 (Bq m ⁻³ WL ⁻¹) C _{pot} (WL)		275 (Bq m ⁻³ WL ⁻¹) C _{pot} (WL)	

T a b l e 5

Equilibrium ratios for radon daughters
and equilibrium factor F for various ventilation rates

Ventilation rate λ _v (h ⁻¹)	Equilibrium ratio x _{a,j} /x _{a,Rn}			Equilibrium factor F
	²¹⁸ Po	²¹⁴ Pb	²¹⁴ Bi	
0.0	1.0	1.0	1.0	1.0
0.1	0.993	0.956	0.913	0.928
0.3	0.978	0.820	0.718	0.784
0.5	0.965	0.729	0.590	0.689
0.7	0.951	0.655	0.530	0.628
1.0	0.932	0.566	0.384	0.526
1.5	0.900	0.473	0.277	0.436
2.0	0.872	0.381	0.195	0.356
3.0	0.820	0.279	0.115	0.269
5.0	0.732	0.173	0.051	0.182
10.0	0.577	0.077	0.013	0.103

Table 5, continued

Ventilation rate λ_v (h^{-1})	Equilibrium ratio $x_{a,j}/x_{a,Tn}$			Equilibrium factor F
	^{216}Po	^{212}Pb	^{212}Bi	
0.0	1	1	1	1
0.1	1	0.395	0.345	0.391
0.3	1	0.179	0.125	0.174
0.5	1	0.116	0.067	0.112
0.7	1	0.086	0.042	0.082
1.0	1	0.061	0.025	0.058
1.5	1	0.042	0.013	0.039
2.0	1	0.032	0.008	0.030
3.0	1	0.021	0.004	0.020
5.0	1	0.013	0.0016	0.012
10.0	1	0.006	0.0004	0.006

Table 6

Published values of radon exhalation rates per unit area [W15]

Location <u>a/</u>	Soil group	Radon exhalation rate per unit area ($mBq\ m^{-2}\ s^{-1}$)	Original ref.
<u>Austria</u>			
Graz	Mountain	20, 9	[K21]
Innsbruck	Mountain	8.6	[Z1]
Innsbruck	Mountain	19	[Z2]
<u>France</u>			
Saclay (80)	Podsollic	15, 14	[S50]
<u>Germany, Fed. Rep.</u>			
Aachen	Podsollic	17	[I12]
<u>Ireland</u>			
Dublin	Podsollic	27	[S51]
<u>Japan</u>			
Osaka (4,4)	Latosolic	3.4, 8.8	[M24]
<u>Philippines</u>			
Manila	Latosolic	11	[W18]
<u>United States</u>			
Socorro, New Mexico (10)	Desertic	34 + 3.4	[W19]
Socorro, New Mexico (6)	Desertic	38 + 11	[P16]
Yucca Flat, Nevada	Desertic	18	[K17]
Lincoln, Massachusetts (10)	Podsollic	50	[K17]
Champaign County, Illinois (472)	Chernozemic	53	[P16]
Argonne, Illinois (8)	Chernozemic	21 + 1.9	[P16]
<u>USSR</u>			
Kirov (36)	Podsollic	15	[M25]
Moscow (6)	Podsollic	3.8	[M25]
Central European Territory (40)	Podsollic	6.9	[S52]
Southwest Kazakhstan (5)	Desertic	5.0	[S52]
Sandy desert (Muyun Kum, Ashkhabad, Dzhusaly) (10,5)	Desertic	4.8, 13	[K22]
The North (Murmansk, Arkhangelsk)(6)	Podsollic	3.8	[K22]
Central European Territory (35)	Podsollic	7.3	[K22]
Leningrad, Moscow, Kaluga reg.	Chernozemic		
The Caucasus (15)	Mountain	11	[K22]
Grozny, Baku, Tbilisi, Adler	Chernozemic		
Middle Asia (10)	Mountain	19	[K22]
Tashkent, Alma-Ata, Frunze	Desertic		
South Urals (5)	Mountain	11	[K22]
Sverdlovsk	Podsollic		
Chelyabinsk	Mountain	11	[K22]
	Podsollic		

Table 7
Radon exhalation from rock and uranium minerals
[19]

Type of rock	Radium activity concentration (Bq kg ⁻¹)	Radon exhalation rate for cores	
		(mBq m ⁻² s ⁻¹)	(μBq m ⁻² s ⁻¹ / (Bq kg ⁻¹))
Leptite a/	79	0.4	5.1
Aptite (1)	460	1.2	2.6
Aptite (2)	280	0.88	3.1
Pegmatite (1)	340	5.3	15.6
Pegmatite (2)	330	8.3	25.2
Uranium minerals			
Sample 1	110000	260	2.4
Sample 2	310000	780	2.5
Sample 3	81000	170	2.1
Sample 4	72000	63	0.9
Sample 5	4500000	190	0.04
Sample 6	240000	7.8	0.03
Sample 7	28000	10	0.36

a/ The radon exhalation rate from a gallery surface was 7.4 mBq m⁻²s⁻¹.

Table 8
Activity concentrations (kBq m⁻³) of ²²⁰Rn and ²²²Rn at different depths
for different pressure conditions
[19]

Nuclide	Depth (m)	Barometric pressure (mb)			
		971	993	1007	1028
²²² Rn	0.1	38	38	16	19
	0.4	47	51	22	24
	0.9	64	67	43	45
²²⁰ Rn	0.1	120	97	90	87
	0.4	130	120	120	140

Table 9
Concentrations (kBq m⁻³) of ²²⁰Rn and ²²²Rn at different depths
for different wind speeds
[19]

Nuclide	Depth (m)	Wind speed at a height of 10m (m s ⁻¹)			
		1.5	3.3	5.7	7.9
²²² Rn	0.1	36	26	20	23
	0.4	46	37	30	28
	0.9	75	58	44	44
²²⁰ Rn	0.1	100	99	74	91
	0.4	120	120	120	150

Table 10

Mean activity concentrations of some building materials

Country	Material	Number of samples	Mean activity concentration (Bq kg ⁻¹)		Ref.	Comments
			²²⁶ Ra	²³² Th or ²²⁸ Th		
Germany, Federal Republic of	Building sand and gravel	50	< 15	< 19	[N2]	Many sources
	Granite	32	100	81	[N2]	Different types
	Bricks (traditional constituents)	109	59	67	[N2]	Different types
	Pumice-aggregate concrete blocks	31	74	81	[N2]	Adequate sampling
	Slag-aggregate concrete blocks	9	152	100	[N2]	Depends on feed materials
	Portland cement	14	< 26	< 19	[N2]	Several sources
	Natural gypsum	23	< 19	< 11	[N2]	Many sources
	Chemical gypsum (phosphogypsum)	33	555	< 19	[N2]	Depends on source of rock
	Red mud bricks	23	281	233	[N2]	Variable composition
Fly ash	28	211	130	[N2]	Many sources	
Hungary	Concrete	95	13	11	[T7]	Average weighted by the relative production
	Brick	176	56	48	[T7]	Average weighted by the relative production
Italy	Lithoid tuff (tufo litoide, Monte Cimino)	-	129	122	[N2]	Commonly used for house building
	Nenfro (a variety of tuff, Tuscania)	-	241	218	[N2]	Wall cladding material
Poland	Fly ash	106	63-610	33-320	[P4]	Typical building material used in Poland
	Slag	42	19-460	22-590	[P4]	
	By-product gypsum	4	26-740	11-44	[P4]	
	Red brick	3	19-22	22-44	[P4]	
	Silicon brick	3	7.4-15	<4-7.4	[P4]	
	Cement	4	7.4-26	11-67	[P4]	
United Kingdom	Soil	5	3.7-19	<4-15	[P4]	
	Granites	7	89	81	[H2]	Inadequate sampling
	Sand and gravel	10	4	7	[O2]	Inadequate sampling
	Cement	6	22	18	[O2]	Inadequate sampling
	Clay bricks	25	52	44	[H2]	Sampling probably inadequate
	White bricks (autoclaved flint and quicklime)	5	4	5	[H2]	Aggregate may vary
	Natural gypsum	73	22	7	[H2]	Adequate sampling
	Lightweight blocks various aggregates	10	59	26	[H2]	Inadequate sampling
	Phosphogypsum from sedimentary ores	60	629	18	[O2]	Depends on ore source
United States (exceptional values)	Phosphate land fill, Florida	-	740	-	[U2]	Estimate depends on geological structure and reclamation procedure
	Gypsum from Florida phosphate rock	-	1221	10	[U3]	Samples from several processing facilities
	Uranium mine tailings	-	4625	-	[C11]	Personal assay of complex situation

Table 11

Mean activity concentrations of some building materials in Nordic countries

(Bq kg⁻¹)

Building material	Country	No. of samples	²²⁶ Ra			²³² Th			Comments	Ref.	
			Min.	Aver.	Max.	Min.	Aver.	Max.			
Brick	Denmark	79	23	42	86	21	34	58	Adequate sampling	[U1]	
	Finland	37	37	80	134	37	62	91	Adequate sampling	[M21]	
	Norway	18		63			74			[S41]	
	Sweden	12	41	96	152	100	127	178	Inadequate sampling	[S42]	
Bricks of limestone	Denmark	5	6	8	11	4	7	11	Adequate sampling	[U1]	
	Finland	3	20	22	25	18	23	29	Inadequate sampling	[M21]	
	Sweden	3	7	10	15	4	8	10	Inadequate sampling	[S42]	
Concrete	Denmark	6	13	16	24	9	13	17	Inadequate sampling	[U1]	
	Finland	12	50	61	80	28	37	42		[M22]	
	Norway	137	11	28	37	21	36	54	Standard deviation 38%	[S41]	
	Sweden	14	32	47	58	56	80	105	Inadequate sampling	[H1]	
Cement	Denmark	6	9	20	30	4	12	21	Adequate sampling	[U1]	
	Finland	9	20	44	84	8.5	22	55	Inadequate sampling	[M21]	
	Norway	4		30			19		Inadequate sampling	[S23]	
	Sweden	16	20	41	168	24	40	81	Adequate sampling	[L11, S42]	
Concrete ballast (gravel, shingle macadam)	Denmark	107	4	19	95	4	13	56	Adequate sampling	[U1]	
	Finland	266	7	34	146	1.6	39	226	Adequate sampling	[M21]	
	Sweden	306	7	48	167	3	72	463	Adequate sampling	[H1]	
Aerated concrete based on sand	Denmark	2		18			10		Inadequate sampling	[U1]	
	Finland	2	45	49	53	31	36	40	Inadequate sampling	[M21]	
	Sweden	24	7	35	130	4	42	155	Adequate sampling	[S42]	
Aerated concrete based on alum shale a/	Denmark	2		670			53		Swedish origin	[U1]	
	Sweden	70	620	1300	2620	30	67	115	Adequate sampling	[S42]	
Aerated concrete based on alum shale b/	Sweden	12	320	466	560	24	30	37	Adequate sampling	[S42]	
Plasterboards	Natural gypsum	Denmark	7	6	10	13	4	4	6	Adequate sampling	[U1]
		Sweden	8	1	4	9	1	1	12	Adequate sampling	[S42]
	Phosphogypsum	Sweden	1		27		65			Inadequate sampling	[S42]
Gypsum	Natural	Denmark	6	4	7	10	4	4	4	Adequate sampling	[U1]
		Finland	1		7		1.5			Inadequate sampling	[M21]
		Norway	2		11		3			Inadequate sampling	[S23]
	Phosphogypsum	Finland	4	24	178	330	3.4	12	22	Inadequate sampling	[M22]
Light-weight aggregate	Denmark	3	36	40	43	37	45	51	Inadequate sampling	[U1]	
	Norway	12		51			56		Inadequate sampling	[S42]	
	Sweden	6	135	170	195	153	164	186	Inadequate sampling	[S42]	
Slag aggregate	Finland	3	88	102	113	32	69	94	Inadequate sampling	[M21]	
	Sweden	2	84	118	151	114	148	182	Inadequate sampling	[U1]	
Insulation material wool of stone or glass	Denmark	5		40			40		Inadequate sampling	[U1]	
	Finland	2	8.9	19	29	4.6	8.9	13	Inadequate sampling	[M21]	
	Sweden	3	11	13	15	15	15	15	Inadequate sampling	[S42]	
Tile	Finland	5	63	78	91	32	46	64	Inadequate sampling	[M21]	
Fly ash	Denmark	10	110	150	210	74	90	160	Adequate sampling	[U1]	
Wood	Finland	2	0.3	0.4	0.5	0.2	0.7	1.2	Inadequate sampling	[M21]	
Clinker	Denmark	13	22	66	108	22	55	73	Inadequate sampling	[U1]	

a/ In production 1929-1975.

b/ In production 1974-1979.

Table 12

Radon concentration in natural gas at the well
[H8, S43, J8, G18, W21]

Location of well	Radon concentration (kBq m ⁻³)	
	Average	Range
Canada		
Alberta	2	0.4-8
British Columbia	18	14 - 20
Ontario	6	0.15-30
Germany, Fed. Rep. of	-	0.04-0.4
Indonesia		
Borneo, Ampa field	-	0.06-0.12
Netherlands		
Slochteren	-	0.044-0.10
Other fields	-	0.14-2
Nigeria		
Niger delta	-	0.034-0.11
United States		
Colorado, New Mexico	0.9	-
Oklahoma, Kansas, Texas	4	0.2-54
Texas Panhandle	-	0.4-19
Colorado	0.9	0.4-2
Project Gasbuggy area	0.6	-
California	-	0.04-4
Kansas	4	-
Wyoming	0.4	-
Gulf Coast (Louisiana, Texas)	0.2	-
California, Louisiana, Oklahoma, Texas	-	0.04-4
North Sea		
Leman field	-	0.07-0.14
Indefatigable field	0.08	-
7 streams supplying		
United Kingdom	0.03	0.01-0.04

Table 13

The relative significance of different radon sources
in a reference house

Source	Radon emission per unit time (kBq d ⁻¹)	Comments
Building materials and soil under the building ^{a/}	60	Exhalation rate $2 \cdot 10^{-3} \text{ Bq m}^{-2} \text{ s}^{-1}$
Water	4	$1 \text{ m}^3 \text{ d}^{-1}$ and 4 kBq m^{-3} , 100 % release
Outside air	10	Radon concentration outdoors 4 Bq m^{-3} , ventilation rate 0.5 h^{-1}
Natural gas	3	
LPG	0.2	

^{a/} In some houses the soil may be the dominating source.

Table 14

Description of the test sites in private and public buildings in Innsbruck, Austria [S21]

Test site No.	Purpose of the test site	Building material	Distance from the ground (m)	Time (d)
1	Control station: cellar	Brick	-1	84
2	Control station: living-room	Brick	4	234
3	Control station: office	Brick	15	58
4	Laboratory	Brick	20	22
5	School storage room	Brick	-1	21
6	School library	Brick	14	25
7	Storage room	Brick, concrete	-2	31
8	School library	Brick, stone	12	24
9	Storage room	Brick	1	25
10	School storage room	Brick	7	18
11	School classroom	Brick	5	18
12	Warehouse	Wood	0	17

Table 15

Activity concentrations in room air of private and public buildings in Innsbruck, Austria [S21]

Test site No.	^{222}Rn (Bq m^{-3})					^{214}Pb (Bq m^{-3})				
	Max.	Min.	$\Delta_{\text{X}}^{\text{a}}/$	Median	Mean	Max.	Min.	$\Delta_{\text{X}}^{\text{a}}/$	Median	Mean
1	276	26.3	81.4	108	115	184	26.3	71.4	98.1	103
2	174	<1.9	115	31.5	32.6	49.6	4.07	26.6	18.9	19.2
3	61.1	<1.9	37.4	29.6	30.7	52.2	8.88	30.0	24.8	25.5
4	34.0	<5.6	13.7	13.0	13.3	27.4	<1.9	17.8	8.88	10.7
5	82.9	22.9	21.8	45.1	46.3	73.3	21.5	25.5	38.9	40.0
6	74.0	15.2	24.8	42.6	43.3	57.0	9.25	21.1	29.2	30.0
7	124	<5.6	72.9	45.9	50.0	81.8	2.22	59.2	30.7	38.1
8	61.1	16.7	17.4	41.4	42.2	29.6	12.2	6.66	18.5	19.6
9	82.5	<5.6	53.3	35.5	37.0	46.6	2.22	25.2	20.0	20.7
10	80.7	42.9	20.4	57.4	58.1	71.4	28.1	30.0	45.5	46.3
11	45.9	6.29	36.3	27.0	27.8	45.1	5.55	34.0	24.4	25.9
12	58.8	11.5	24.8	32.6	33.7	54.0	2.96	30.0	21.1	22.9

Test site No.	^{220}Rn (Bq m^{-3})					^{212}Pb (mBq m^{-3})				
	Max.	Min.	$\Delta_{\text{X}}^{\text{a}}/$	Median	Mean	Max.	Min.	$\Delta_{\text{X}}^{\text{a}}/$	Median	Mean
1						5720	1500	1420	3350	3470
2						4110	755	2180	2000	2090
3						6560	2230	2340	4560	4700
4	28.1	<3.3	17.0	7.40	7.77	947	455	363	692	703
5	16.3	<3.3	11.1	8.51	8.51	4590	481	1910	2450	2850
6	10.0	5.55	52.2	37.0	40.7	3620	1340	555	1880	2180
7	35.9	<3.3	14.1	14.8	18.5	3300	747	1520	1860	1940
8	74.0	23.3	10.0	39.2	41.4	4920	2890	492	3570	3600
9	32.6	<3.3	17.8	10.4	11.1	2780	344	2080	1890	1900
10	35.5	11.8	16.7	19.2	20.7	3130	1580	718	2280	2320
11	8.88	<3.3	4.44	3.7	3.7	2600	710	881	1670	1750
12	33.7	7.77	13.0	19.2	20.7	507	59.2	259	285	329

$\Delta_{\text{X}}^{\text{a}}$ is the largest observed increase of the activity concentration within 24 hours.

Table 16

Correlation between radon and thoron concentration in houses
and meteorological variables
[521]

Meteorological variable	Observation time of meteorological change	Sign of regression coefficient
Change of barometric pressure	24 h	-1
Change of barometric pressure	10 h	-1
Change of soil temperature at 50 cm depth	24 h	+1
Change of daily mean temperature in the open atmosphere	24 h	+1
Change of daily mean wind speed	24 h	-1
Temperature gradients between 574 and 918 m above sea level at 7 h		+1
Change of relative humidity in the open atmosphere	7 h	-1
Change of temperature in the open atmosphere	7 h	+1
Change of daily range of temperature in the open atmosphere	24 h	-1

Table 17

Equilibrium factors in dwellings in Finland 1977-1978
[M20]

Type and number of dwellings	Radon concentration (Bq m ⁻³)			Equilibrium factor F
	Max.	Mean	Min.	
Block of flats, concrete (15)	140	48	7.4	0.45
Detached houses, concrete (2)	190	130	48	0.30
Block of flats, brick (7)	56	26	7.4	0.63
Detached houses, brick (5)	85	33	15	0.45
Detached houses, wood (6)	70	30	7.4	0.43

T a b l e 18

Dose equivalents per unit inhaled
potential alpha energy of ^{212}Pb and ^{212}Bi
assuming an AMAD of 0.2-0.3 μm and a biological half-time of residence
in lung for the transfer from lung to blood in the range of 0.2-0.5 day
[J13]

Organ or tissue	Dose equivalent per unit inhaled potential α -energy (Sv J^{-1})		
	^{212}Pb	^{212}Bi	
Lung	Bronchial basal cells (B)	3 - 5	9 - 10
	Pulmonary region (P)	1.2 - 2.5	4 - 5
	Total lung average (Lu)	1.5 - 3	5 - 5.5
Red bone marrow	0.25 - 0.15	0.01 - 0.005	
Bone lining cells	3 - 2	-	
Kidney	2 - 1.2	0.9 - 0.4	
Liver	0.4 - 0.25	0.02 - 0.01	
Spleen	0.08 - 0.05	0.04 - 0.02	
Other tissues	0.03 - 0.02	0.01 - 0.005	
Effective dose $w_B=w_P=0.06$	0.6	0.9	
equivalent per unit inhaled potential alpha energy $w_{Lu} = 0.12$	(0.54 - 0.63)	(0.84 - 0.93)	
	0.5	0.7	
	(0.47 - 0.54)	(0.66 - 0.70)	
Recommended effective dose equivalent per unit inhaled potential alpha energy for mixtures of ^{212}Pb and ^{212}Bi		0.7	

T a b l e 19

Dosimetric coefficients relative to inhalation
of radon daughters by miners

Quantity	Unit	^{222}Rn daughters a/	^{220}Rn daughters b/
$H_{\text{eff}}/I_{\text{pot}}$	Sv/J	2	0.7
$H_{\text{eff}}/C_{\text{pot}}$	$\text{Sv}/(\text{J h m}^{-3})$	2.4	0.84
	mSv/WLM	8.4	2.9
$H_{\text{eff}}/\tilde{x}_{\text{eq}}$	$\eta\text{Sv}/(\text{Bq h m}^{-3})$	14	64

a/ For inhalation of ^{222}Rn gas itself the effective dose equivalent per unit activity inhaled is about $1.5 \cdot 10^{-10} \text{ Sv Bq}^{-1}$.

b/ The values refer to the inhalation of mixtures of $^{212}\text{Pb}+^{212}\text{Bi}$. For inhalation of $^{220}\text{Rn}+^{216}\text{Po}$ the effective dose equivalent per unit inhalation intake is about $1 \cdot 10^{-10} \text{ Sv Bq}^{-1}$.

Table 20
Breathing rates used for dose calculations
(12)

Location	Period	Breathing rate	Activity
Indoors (19 h)	5.5 h	20 l min ⁻¹	light activity
	8 h	7.5 l min ⁻¹	resting
	5.5 h	12.5 l min ⁻¹	intermediate
Outdoors (5 h)	2 h	20 l min ⁻¹	
	3 h	12.5 l min ⁻¹	
Indoor total		15 m ³ d ⁻¹	
Outdoor total		5 m ³ d ⁻¹	
Grand total		20 m ³ d ⁻¹	

Table 21
Dosimetric coefficients relative to indoor
and outdoor inhalation of radon daughters by members of the public^{a/}

Quantity	Unit	²²² Rn daughters		²²⁰ Rn daughters b/	
		Indoors	Outdoors	Indoors	Outdoors
H _{eff} /I _{pot}	Sv/J	2.0	3.0	0.7	0.7
H _{eff} /C _{pot}	Sv/J h m ⁻³	1.6	3.0	0.55	0.7
	mSv/WLM	5.5	11	1.9	2.5
H _{eff} /x _{eq}	μSv/Bq h m ⁻³	8.7 · 10 ⁻³	17 · 10 ⁻³	40 · 10 ⁻³	53 · 10 ⁻³

a/ See Table 19 for the values relative to inhalation of ²²²Rn and ²²⁰Rn+²¹⁶Po.

b/ Inhalation of mixtures of ²¹²Pb+²¹²Bi.

Table 22
Annual effective dose equivalents per unit of activity concentration,
or of potential alpha energy concentration,
for daughters of ²²⁷Rn and of ²²⁰Rn
and for outdoor and indoor exposure

(Calculated with an annual breathing volume
V_{in} = 5475 m³ indoors and V_{out} = 1825 m³ outdoors)

Inhaled	Contribution	H _{eff} /C _{pot}	H _{eff} /x _{eq}
		$\frac{\text{Sv}}{\text{J m}^{-3}}$	$\frac{\text{mSv}}{\text{Bq m}^{-3}}$
²²² Rn daughters	Indoors	1.1 · 10 ⁴	0.061
	Outdoors	5.5 · 10 ³	0.031
²²⁰ Rn daughters	Indoors	3.8 · 10 ³	0.29
	Outdoors	1.3 · 10 ³	0.10

a/ Inhaled mixtures of ²¹²Pb and ²¹²Bi.

Table 23

^{222}Rn and ^{212}Pb concentrations in outdoor air

Location	Mean value (Bq m^{-3})		Ref.
	^{222}Rn	^{212}Pb	
Austria	7.0		[S22]
Bolivia	1.5		[L8]
Finland	2.3		[M5]
	3.5		[M5]
	3.8		[M5]
	2.0		[M5]
	2.7		[M5]
	3.6		[M5]
France	9.3		[F4]
Germany, Fed.Rep. of	2.6		[16]
India	3.7	0.11	[R4]
Japan	2.1		[L8]
Peru	1.5		[L8]
Philippines	0.3	0.01	[B6]
Poland	3.3		[P19]
Soviet Union	6.3		[M2]
	2.6	0.11	[M1]
	3.3	0.12	[M1]
	2.2	0.09	[M1]
United Kingdom	3.3		[H6]
United States (continental)			
Chicago	1.6	0.1	[B6]
	1.5	0.07	[B6]
Washington	2.9	0.07	[B6]
	4.4		[L8]
San Diego	0.1	0.007	[B6]
San Francisco	0.6	0.002	[B6]
Seattle	0.1	0.004	[B6]
Memphis	1.0	0.7	[B6]
New York City	4.8		[G10]
	3.7		[T3]
Chester, N.J.	7.9		[F5]
Cincinnati	9.6		[G12]
New Mexico	8.9		[W6]
Puerto Rico	0.005	0.001	[B6]
Alaska	0.2	0.001	[B6]
North Africa	0.5	0.008	[B6]
Norwegian Sea	0.2		[L5]
Pacific Islands			
Hawaii	0.2		[L3]
	0.05		[B6]
Marshall Islands	0.02		[B6]
Caroline Island	0.02		[B6]
Mariannas (Guam)	0.05		[B6]
Samoa	0.08	0.004	[B6]
Indian Ocean	0.07		[S6]
North Atlantic	0.2		[S6]
South Pacific	0.07		[S6]

Note: The values given in this table should not be considered to be mean values for the entire countries.

Table 24

Normalized annual effective dose equivalent from outdoor exposure
(20 % of the total time) to radon daughters from radon released from coal
and geothermal plants and from tailings

Distance (km)	Normalized annual effective dose equivalent					
	Coal power plant	Geothermal power plant	Tailings			
			Area = 10^4 m^2		Area = 10^6 m^2	
			$\text{nSv}(\text{GW}(\text{e}) \text{ a})^{-1}$	$\mu\text{Sv}(\text{GW}(\text{e}) \text{ a})^{-1}$	μSv	$\mu\text{Sv}(\text{GW}(\text{e}) \text{ a})^{-1}$
0.5	0.88	5.5	2.7	11	21	0.9
1	1.5	9.4	1.1	4.3	14	0.6
2	1.3	7.9	0.3	0.08	8.8	0.4
5	0.57	3.4	0.08	0.3	4.6	0.2
10	0.31	1.8	0.03	0.1	2.4	0.09

Table 25

Radon concentration in water

Location	Number of wells with radon concentration in water				Radon concentration in water		Ref.
	< 37 kBq m ⁻³	37-370 kBq m ⁻³	0.37-3.7 MBq m ⁻³	3.7-37 MBq m ⁻³	Maximum kBq m ⁻³	Average kBq m ⁻³	
<u>Austria</u> Salzburg					7	1.5	[S22]
<u>Finland</u> Helsinki and Vantaa	4	12	65	29		1200	[C3]
Other areas	11	34	30	7	45000	280	[C3]
<u>Italy</u>	41	16	2	-		80	[H3]
<u>Sweden</u>	155	17	-	-	150	19	[K14]
<u>United States</u>							
Aroostock, Maine	13	19	-	-	200	48	[H10]
Cumberland, Maine	1	6	7	2	5800	1000	[H10]
Hancock, Maine	1	3	11	1	4600	1400	[H10]
Lincoln, Maine	3	6	10	1	1600	560	[H10]
Penobscot, Maine	-	10	6	-	2400	540	[H10]
Waldo, Maine	-	5	9	-	3100	1100	[H10]
York, Maine	-	6	9	-	2200	670	[H10]
All 7 counties	18	55	52	4	5800	660	[H10]
North Carolina	84	117	10	-	1700	100	[S1]

Table 26

Distribution of radon concentration
in drinking water in Finland
[A7, A11, C4, K18]

Water specification	Number of persons (million)	Radon concentration (kBq m ⁻³)	
		Mean	Maximum
Waterworks	3.18	25	1600
Dug wells	1.39	59	1600
Drilled wells	0.15	630	44000
Drilled wells in Helsinki area	0.02	1200	
Weighted average		56	

Table 27

Radon and radon daughter concentrations
in Canadian homes
[N6]

Location	Radon concentration (Bq m ⁻³)		Equilibrium equivalent radon concentration (Bq m ⁻³)	
	Geometric mean	Geometric standard deviation	Geometric mean	Geometric standard deviation
Calgary, Alberta	11	3.6	7.0	2.3
Charlottetown, Prince Edward Island	15	5.3	6.7	2.6
Fredericton, New Brunswick	24	4.0	12	2.9
Halifax, New Brunswick	-	-	11	3.1
Montreal, Quebec	11	3.3	5.2	2.5
Quebec, Quebec	10	3.8	4.8	2.7
Saint John, New Brunswick	10	5.7	6.7	3.0
Sherbrooke, Quebec	13	5.4	8.5	3.3
St. John's, Newfoundland	11	4.4	5.6	2.7
St. Lawrence, Newfoundland	33	6.8	6.3	4.6
Sudbury, Ontario	21	4.0	13	3.0
Thunder Bay, Ontario	20	4.5	9.3	2.6
Toronto, Ontario	11	2.8	6.7	2.6
Vancouver, British Columbia	5.2	3.0	3.3	2.0
Winnipeg, Manitoba	51	4.1	26	3.4
Brandon, Manitoba	30	4.8	16	2.7
Regina, Saskatchewan	47	3.6	19	3.1
Saskatoon, Saskatchewan	16	4.3	16	3.3
Edmonton, Alberta	16	4.6	16	3.3

Table 28

Equilibrium equivalent radon concentration in houses
in Canada suspected of having enhanced concentrations
[K9]

Location	Number of houses	Average equilibrium equivalent radon concentration (Bq m ⁻³)
Port Hope (Radium/uranium refinery)	2961	11
Cobourg (reference area) (Spring)	106	5.2
(Autumn)	97	5.6
Uranium City (Uranium mining)	632	48
Elliot Lake (Uranium mining)	1921	30
Bancroft area (Uranium mining)	1162	26
Deloro (Metallurgical operations)	68	22

Table 29

Radon and radon daughter concentrations in indoor air
and radon concentration in water supply in Canada
[M7]

Area	Location	Air			Water (MBq m ⁻³)	Air-to- water concent- ration ratio (10 ⁻⁴)
		Radon (Bq m ⁻³)	EEC a/ (Bq m ⁻³)	F b/ (10 ⁻⁴)		
Harriets- field	Trailer	< 18.5	7.4	> 0.4	5.8	< 0.03
	Trailer	118	37	0.3	6.1	0.2
	Trailer	< 22	7.4	> 0.3	5.6	< 0.04
	Trailer	< 74	26	> 0.3	5.8	< 0.1
	Trailer	152	30	0.2	6.2	0.2
School:	Washroom	93	11	0.1	5.5	0.2
	Library	< 18.5	11	> 0.6	5.5	< 0.03
Big Acres	Basement	< 18.5	7.4	> 0.4	5.5	< 0.03
Sheldrake	Basement	126	15	0.1	4.7	0.1
Hubley Lake	Basement	81	38	0.5	1.6	0.5
	Kitchen	26	15	> 0.6		< 0.2
Five Islands Lake	Boiler room	< 44	3.7	> 0.1	3.6	< 0.1
	Transformer	< 22	3.7	> 0.2		< 0.1
Woodland	Basement	707	93	0.1	14	0.5
	Kitchen	240	93	0.4		0.2
Highland Park	Basement	56	11	> 0.2	7	0.08
	Living room	110	7.4	0.1		0.2
West Timberlea	Laundry room	120	19	0.2	12	0.1

a/ Equilibrium equivalent concentration of radon.

b/ Equilibrium factor.

Table 30

Radon and radon daughter concentrations in indoor air
due to warm water showers of 7 minutes

The radon concentration in water was 4.4 kBq m⁻³.
[M7]

Bathroom	Time elapsed (min)	Concentration (Bq m ⁻³)		Distri- bution ratio (10 ⁻⁴)
		Radon	Radon daughters	
Before shower operation		19	7.4	< 0.04
During shower operation	0	148	-	0.3
	1	850	-	1.9
	2	1040	-	2.4
	3	1890	-	4.3
	4	2070	-	4.7
	5	2740	-	6.2
	8	3520	-	8.0
	After shower operation	22	3100	2400
	32	2660	110	
	99	148	26	

T a b l e 31

Radon concentration in air in Finnish dwellings caused by radon in water
[A3]

House no.	Radon concentration										
	Water (MBq m ⁻³)	Air (kBq m ⁻³)									
		Rooms and situations where much water is used (showers)				Rooms and situations where little water is used (cooking)				Living rooms (no water)	
		Room or situation number				Room or situation number					
	1	2	3	4	1	2	3	4	5		
1	5.2	4.2	4.0	1.2	1.5	0.81	1.2	5.8			0.34
2	7.4										0.53
3	5.8										0.093
4	3.7	2.1	22			0.93	0.59				0.23
5	5.3	3.9	2.1	13		1.6	4.6				0.30
6	3.0	15	9.2			6.7	0.67	0.85	9.3		0.23
7	4.2	2.6	2.5	2.4		0.19	0.41	0.41			0.13
8	17	39	29			5.9	27				0.36
9	7.6	6.0				0.81					0.31
10	5.0	23	10			4.7	3.4	1.4	6.0	3.0	0.31
11	8.0	11				6.8	0.3	1.0			0.20
12	9.2	10				0.74	1.8	0.26	0.30	2.9	0.13
13	13	19	22			8.7	13		4.2	3.3	0.77
14	0.48	0.22	2.7			0.89					0.019
15	1.5	3.2	1.8			0.037	0.037	0.11			0.059
16	1.0	3.8	2.1			0.15	0.56	2.6	0.78		0.041
17	0.15	0.85	0.67			0.074		0.037	0.074		0.037
18	2.5	3.6				0.56	0.44				0.11
19	1.9										0.056
20	0.52										0.078
Average	(5.1 ± 4.3)	8.5 ± 9.7				2.9 ± 4.7					0.22 ± 0.19

T a b l e 32

Equilibrium equivalent concentration of radon and thoron
in dwellings in the Federal Republic of Germany

Ranges are given between parentheses.
[W14]

Building material	Equilibrium equivalent concentration (Bq m ⁻³)	
	Radon	Thoron
Concrete based and mixed building material	10 (3.7-25)	0.44 (0.19-2.2)
Sandstone	5.6	0.30 (0.11-0.48)
Brick	4.8 (3.7-6.3)	0.70 (0.30-1.1)
Aerated concrete	3.3 (2.6-4.1)	0.11
Light-weight building material	3.7 (3.0-4.8)	0.19 (0.11-0.26)
Weighted average	8.1 (2.6-25)	0.37 (0.11-2.2)

T a b l e 33

Radon concentration in dwellings
in Norway 1977-1978
[S24]

Type of dwelling	Radon concentration (Bq m ⁻³)		
	Min.	Mean	Max.
Wood	7.4	48	140
Concrete	7.4	74	250
Brick	11	37	210

T a b l e 34

Radon and radon daughter concentrations in flats in Poland
(G13)

Location and building material	No. of flats	Maximum concentration of radon (Bq m ⁻³)	Equilibrium equivalent radon concentration (Bq m ⁻³)	
			Range	Weighted average
Warsaw Different types of concrete	11	2.2-28	1.1-14	7.0
Lublin Concrete and gypsum	39	15	0.4-7.8	2.0
Concrete, siporex and gypsum	32	23	0.4-13	3.7
Wood	9	2.1-9.4	1.0-4.8	2.9
Brick	29	1.2-33	0.63-17	8.9
Limestone	3	2.4-19	1.2-8.9	4.8
Clay	6	2.3-17	1.2-8.5	4.8
Clay and brick	4	1.6-17	0.81-8.5	4.8
Slag and cement	12	4.4-52	2.3-27	15
Clay, slag and cement	2	9.3-16	4.8-8.5	6.7
Brick and siporex	5	2.3-14	1.2-7.8	4.4
Concrete and siporex	7	1.5-7.4	0.78-4.1	2.4
Brick, slag and cement	10	2.9-28	1.3-14	7.8
Weighted average of all flats				5.6

T a b l e 35

Average activity concentration of radon and radon daughters in air, absorbed dose rate in air from gamma radiation and air exchange rates in houses built of aerated concrete based on alum shale a/
(S45)

Date ready for moving in	Number of houses	Building material	Concentration		Air exchange rate _a (h ⁻¹)	Absorbed dose rate in air from gamma-radiation (nGy h ⁻¹)
			Radon (Bq m ⁻³)	Radon daughters (Bq m ⁻³)		
1969	9	Entirely built of alum shale aerated concrete	780 (535-1160)	410 (190-770)	0.31 (0.21-0.43)	635 (580-690)
1962	7	All walls built of alum shale aerated concrete	490 (320-690)	185 (80-250)	0.41 (0.24-0.55)	380 (350-440)
1968	9	as above	370 (190-490)	170 (75-280)	0.49 (0.34-0.61)	425 (410-440)
1967	7	As above	590 (175-820)	245 (45-355)	0.27 (0.17-0.49)	540 (480-600)

a/ The lowest and highest values calculated as averages for a house are given in parentheses.

T a b l e 36

The concentrations of radon and daughter products
in 63 dwellings in a Swedish town
 [E2]

Building materials in the walls and ventilation system a/	Air exchange rate (h ⁻¹)		Radon (Bq m ⁻³)		Equilibrium equivalent radon concent- ration (Bq m ⁻³) b/		
			I	II	I	II c/	
<u>Multi-family houses</u>							
Concrete, ME	0.3-0.6	low	59	48	22	11	
		average	170	140	70	52	
		high	590	780	260	310	
Concrete and sandbased aerated concrete, ME	0.5-0.9	low	37	26	13	9	
		average	89	85	31	23	
		high	150	140	48	41	
Concrete and sandbased and alum shale based aerated concrete, ME	0.4-0.8	low	74	93	22	11	
		average	180	160	52	44	
		high	440	410	150	140	
<u>Single-family houses</u>							
Facade brick (sandstone), wood construction, ME	0.4-0.7	low	n d/			8	
		average	56			22	
		high	140			38	
Wood, cellar of alum shale based aerated concrete, ME	0.4-0.7	low	n d/			4	
		average	200			81	
		high	370			140	
Wood, no cellar, ME	0.8	low	100			33	
		average	100			37	
		high	100			41	
Alum shale based aerated concrete, DV	0.2-0.5	low	150			67	
		average	270			120	
		high	410			190	
Facade brick (clay), wood construction, DV	0.1-0.2	low	220			82	
		average	410			170	
		high	560			310	

a/ ME is a mechanical exhaust ventilation system, DV is a draught ventilation system in the houses studied with a fan over the cooker.

b/ The radon daughter concentrations were measured

c/ I and II mean the first and second phases between which the ventilation systems were adjusted

d/ n ≤ minimum detectable value, 26 Bq m⁻³ for radon.

T a b l e 37

Average radon concentrations
in Swedish dwellings
 [S35]

Houses existing in	Radon concentration (Bq m ⁻³)		
	Apartment houses	Detached houses	Average
1950	43	18	29
1975	89	48	71
1985 a/	88	51	73

a/ Predicted values.

T a b l e 38

Equilibrium equivalent radon concentrations in houses
built on phosphate land in the United States
[U8]

Structure	Type of land	Equilibrium equivalent concentration ^{a/} (Bq m ⁻³)		
		Geometric mean	Range	
Slab-on-grade	Undisturbed	11	3.7-37	
	Unmined, near-surface radioactive deposits and fill	Tailings	70	11-170
		Lower activity overburden	30	7-140
		Higher activity overburden and debris	30	15-67
		160	70-520	
Crawl space and mobile homes	Undisturbed	11	3.7-37	
	Reclaimed	22	3.7-52	

a/ The weighted average was 37 Bq m⁻³, which is estimated to be 26 Bq m⁻³ above normal background.

T a b l e 39

Equilibrium equivalent radon concentrations
in houses in Colorado, United States
[C11]

Number of houses	Equilibrium equivalent concentration (Bq m ⁻³)
15	> 196
26	52-196
5	19- 48
1	< 19

T a b l e 40

Equilibrium equivalent concentrations of radon and annual effective dose equivalents
for dwellings of different types in various countries

The equilibrium factor is assumed to be 0.5.

Country or area	Structure	Equilibrium equivalent concentration (Bq m ⁻³)	Annual effective dose equivalent (mSv)	Ref.
Austria	Average for Salzburg	12	0.7	[S22]
Canada	Typical Canadian homes	17	1.0	[M6]
Denmark	Basement room, thick structured elements	4.8	0.3	[N2]
Finland	Flats other than groundfloor	17	1.0	[N2]
Germany, Fed. Rep.	Average for 32 houses	8.1	0.5	[W14]
Hungary	Isolated rooms	20	1.2	[N2]
	As specified in Table 28, Annex B [U6]	120	7.3	[U6]
Norway	Flats other than groundfloor	11	0.7	[N2]
	Average value	26	1.6	[S24]
Poland	Average value	6-17	0.4-1.0	[G13, B16, B17, U6]
Sweden	Average value	60	3.7	[R15]
United Kingdom	Single-family houses	15	0.9	[N2]
	Average value	13	0.8	[C6]
United States	New Jersey and New York and as specified in Table 28, Annex B [U6]	15	0.9	[U6]
USSR	Flats other than groundfloor	4.8	0.3	[N2]
	Single-family houses and groundfloor flats	16	1.0	[N2]
Several countries	Mainly masonry houses and apartments	18	1.1	[U6]

T a b l e 41

Radon and thoron daughter concentrations in different rooms
in the United Kingdom
[C6]

Equilibrium equivalent concentration (Bq m ⁻³)		Radon-to-thoron equilibrium equivalent concentration ratio
Radon	Thoron	
14	0.33	42
0.7	0.08	9
41	1.2	34
13	0.12	110
54	0.33	160
91	0.41	220
89	0.44	200
6.4	0.43	15

T a b l e 42

Concentrations of, and exposure to, radon daughters in uranium mines

Country	Year	Average potential alpha energy concentration (WL)	Average annual potential alpha energy exposure (WLM)	No. of miners	No. of miners exceeding 4 WLM a/	Ref.
France	1971	0.18				[U6]
	1972	0.17				
	1973	0.18				
	1974	0.13				
	1975	0.11				
	1976					
	1977					
	1978			2.0	1284	~ 140
1979			1.4	1503	51	[B20]
United States	1975	0.71	5.68	~ 5000		[R6]
	1976	0.58	4.64	~ 5000		
	1977	0.51	4.08	~ 5000		
Italy	1975	< 1				[S4]
Canada	1978					[A6]
	1 Leaching		0.38	630		
	4 Underground		0.74	3690		
	1 Open pit		0.41	276		
	1978		0.72	4535	9	b/
1979		0.74	6883	1		
Argentina	Underground 1977-79		2.4	286-379		[P18]
	1980		2.4	95	0	
	Open pit 1980		0.12	285	0	

a/ The maximum permissible exposure in many countries.

b/ Data from the National Dose Registry in Canada.

Table 43

Estimated potential alpha energy exposure
of different categories of mine workers
in the Jaduguda underground mines, India
[R13, K19, A12]

Year	Estimated potential alpha energy exposure (WLM)		
	Drilling crew	Mucking crew	Others
1965	4.9 ± 2.6	2.1 ± 1.0	1.7 ± 1.0
1966	2.3 ± 1.2	3.5 ± 1.8	1.2 ± 0.9
1967	2.0 ± 1.1	5.2 ± 2.7	1.6 ± 1.1
1968	3.8 ± 2.0	3.2 ± 1.6	2.3 ± 1.5
1969	4.1 ± 2.2	6.5 ± 3.4	2.4 ± 1.7
1970	2.1 ± 1.1	3.0 ± 1.4	0.7 ± 0.5
1971	1.7 ± 0.9	2.0 ± 1.1	1.1 ± 0.8
1972	0.7 ± 0.6	1.6 ± 1.4	1.4 ± 1.3
1973	0.6 ± 0.3	0.6 ± 0.3	0.7 ± 0.5
1974	1.6 ± 0.6	5.5 ± 5.0	2.0 ± 1.6
1975	2.2 ± 0.7	2.3 ± 1.9	3.5 ± 1.7
1976	5.5 ± 4.4	2.5 ± 1.1	0.7 ± 0.1
1977	1.6 ± 0.6	1.7 ± 0.7	1.4 ± 0.7
1978	0.8 ± 0.2	1.4 ± 0.7	
1979	2.6 ± 1.0	2.1 ± 0.6	1.7 ± 0.3

Table 44

Concentrations of, and exposure to, radon daughters in non-uranium mines^{a/}

Country	Year	Average potential alpha energy concentration (WL)	Annual potential alpha energy exposure (WLM)	No. of miners /mines	No. of miners exceeding 4 WLM	Ref.
Finland	1972-1974	0.2-0.4	0.38	1300/23		[A5]
	1975-1977			1370/16	0	
Italy	1975	0.01-0.6		2500/16	~ 75	[S4]
Norway	1972	0.07	0.64	1870/33		[S54, S55]
	1980	0.05	0.45	1380/23		
Poland	1970					[D3]
Copper		1-2				
Iron		1				
Pyrite		4				
Phosphate		0.8				
Zinc and lead		0.9				
Baryte		0.2				
Coal		0.1				
South Africa	1973		1.7	320000		[U6]
Sweden	1970		4.8	4800/5	2000	[U6]
	1974		2.1	4600/50	360	[U6]
	1975		1.9	5300/45	270	[S13]
	1976		1.7	5300/46	225	[U6]
	1977		1.6	5200/45	475	[S13]
	1978		0.9	5300/47	270	[S13]
	1979		0.7	4400/35	0	[E1]
	1980		0.7	4400/35	0	[S56]
United Kingdom	1968	0.01 ^{b/}	2-3 ^{c/}	220000/420		[O4]
	1976			2000/80	560	[O4]
	National coal	1981	0.12	185200		[O8]
	Private coal	1981	0.24	1500		[O8]
	Other than coal	1981		2.60	2346/108	94
United States	1975	0.31				[R6]
	1976	0.22				
	1977	0.12			/163	

a/ If not otherwise noted, the mines are iron, zinc, lead, copper or gold mines.

b/ In reference [O4], this value is called "typical" for large nationalized coal mines.

c/ Based on measurements in about 80 % of all non-coal mines.

T a b l e 45

Concentration of, and exposure to, radon daughters
in working places other than mines

Country Kind of room or working place	Average potential alpha energy concentration (WL)	Annual potential alpha energy exposure (WLM)	Number of workers/ working places	Ref.
<u>Austria</u> Public baths (Badgastein)	0.5-0.9	6-11 <u>b/</u>	>100	[P7]
<u>Hungary</u> 3 caves	0.45 <u>a/</u>	5 <u>b/</u>	/3	[R9]
<u>Italy</u> 20 spas	0.001	0.02-24 (18 spas < 1 WLM)	/20	[S5]
<u>Japan</u> 2 caves	0.8 <u>a/</u>			[M23]
<u>Sweden</u> Tunnels for water		1.1	11/1	[S13]
Tunnels for cables		0.45	34/2	[S13]
Defence installations		1 - 4 <u>c/</u>		[S13]
Hydro-electric power stations	~ 0.1-1		200/78	[S13]
<u>United States</u> 6 caves	0.3 - 1	0.1 - 1.5	/6	[Y1,S46]

a/ An equilibrium factor F of 0.5 is assumed.

b/ A fictitious value. An annual occupancy of 2000 h is assumed.

c/ For about 3 % of the staff.

T a b l e 46

Measured values of the concentrations of ^{220}Rn , ^{212}Pb and ^{212}Bi at a gas mantle factory
[D6]

Time of measurement (h)	Location	^{212}Pb concent- rations (Bq m ⁻³)	^{212}Bi concent- rations (Bq m ⁻³)	Ventil- ation rate in room air (h ⁻¹) <u>a/</u>	^{220}Rn Inferred concentration (Bq m ⁻³)	^{220}Rn Measured concentration (Bq m ⁻³)	^{220}Rn inferred ^{220}Rn measured	Ratios of the measured concentrations ^{220}Rn : ^{212}Pb : ^{212}Bi
First day								
10.50	Impregnating room	270	35	63	7400	22000	0.3	100:1.2:0.75
11.20	Impregnating room	150	23	140	8900	13000	0.7	100:1.1:0.18
12.10	Drying room	140	21	140	8500	6300	1.4	100:2.2:0.33
12.20	Drying room	130	21	33	7400	5900	1.2	100:2.3:0.36
12.40	Drying room	140	19	160	9300	4800	1.9	100:2.8:0.38
12.50	Drying room	110	13	210	10000	5200	1.9	100:2.2:0.24
15.25	Impregnating room	220	28	170	16000	18000	0.9	100:1.2:0.16
15.40	Impregnating room	290	31	210	26000	26000	1.0	100:1.1:0.12
Second day								
11.00	Impregnating room	270	48	120	14000	16000	0.8	100:1.7:0.19
11.25	Thorium nitrate store	100	48	30	1500	3700	0.4	100:2.8:1.3
11.40	Thorium nitrate store	70	22	56	1700	2000	0.8	100:3.5:1.1
12.00	Drying room	130	26	100	5900	3000	2.0	100:4.5:0.89
12.15	Drying room	200	29	150	13000	5600	2.3	100:3.6:0.53

a/ Inferred from the ^{212}Pb and ^{212}Bi concentrations.

T a b l e 47

Typical variations of the potential alpha energy concentration of thoron daughters at different locations in a thorium factory in Trombay, India (1974)
[010]

Location	WL
Evaporator room	0.04
Area around mother liquor tank	0.12
Filling room	0.13
Crushing room	0.40
Area around hydroxide tanks	0.06
Drying room	0.24
Area around chloride filter presses	0.10
Laboratory	0.16
Area around tanks	0.07

T a b l e 48

Increase in collective effective dose equivalents caused by radon daughters per unit of electrical energy conserved by reduction of the ventilation rate (to a rate not less than 0.5 h^{-1}) for Swedish houses built up to 1970

Type of house and building material	Normalized collective effective dose equivalent ($10^3 \text{ man Sv (GW(e) a)}^{-1}$)		
	Multi-family houses	Single-family houses	Weighted average
Wooden house	2.8	1.4	1.8
Wooden house, cellar walls of concrete based on alum shale	-	11	11
Brick/concrete	8.4	5.7	7.0
Sand based aerated concrete	-	2.9	2.9
Concrete based on alum shale	20	7.5	14
Weighted average	8.7	2.2	5.6

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