SOURCES AND EFFECTS OF IONIZING RADIATION

United Nations Scientific Committee on the Effects of Atomic Radiation

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NOTE

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ANNEX B

Exposures from natural radiation sources

CONTENTS

INTRODUCTION

1. The exposure of human beings to ionizing radiation from natural sources is a continuing and inescapable feature of life on earth. For most individuals, this exposure exceeds that from all man-made sources combined. There are two main contributors to natural radiation exposures: high-energy cosmic ray particles incident on the earth's atmosphere and radioactive nuclides that originated in the earth's crust and are present everywhere in the environment, including the human body itself. Both external and internal exposures to humans arise from these sources. These exposures were reviewed in previous reports of the Committee, the most recent being the UNSCEAR 1993 Report [U3].

2. In assessing exposures to the natural radiation background, the Committee has considered the properties of the sources and the transport of both radionuclides and radiation in the environment. Estimates have been made of typical exposures to the world population and the range of the components of such exposures under various environmental conditions, and note has been taken of the unusually high natural radiation exposures that occur in some locations. This information has been combined with relevant dosimetric quantities to estimate the absorbed doses in tissues and the effective doses from the various sources of exposure.

3. In this Annex, the Committee continues its general review of the various components of the natural radiation background. To broaden the database, an attempt has been made to gather representative levels of exposure in as many countries as possible. Many scientists and representatives of national institutions have responded to the questionnaire on natural radiation exposures, UNSCEAR Survey of Natural Radiation Exposures, which was widely distributed by the Committee. Respondents to the questionnaire are listed in Part A of the References. The Committee acknowledges with appreciation their useful contributions to its work.

4. The database on natural radiation exposures has become extensive enough to allow quite detailed analysis. For example, the distributions of populations within various dose intervals from the different components of exposure can be examined within and between countries. The processes giving rise to the exposures can be better described and the time and geographic variations more accurately evaluated, allowing some issues to be addressed in greater detail. There remain, however, some questions that are not yet satisfactorily resolved. For example, there are difficulties in evaluating cosmic ray exposures in aircraft because of the complex neutron and ionizing radiation fields, and the dosimetry of inhaled radon is complicated by the complexities and variations of the interacting factors and processes involved.

5. Many exposures to natural radiation sources are modified by human practices. In particular, natural radionuclides are released to the environment in mineral processing and uses, such as phosphate fertilizer production and use and fossil fuel combustion, causing enhanced natural radiation exposures. In a few cases, for example, by paving roads or building houses over water, radiation exposures may be decreased, but these seem to be rather isolated cases. The general topic of enhanced exposures from natural radiation sources was considered in detail in the UNSCEAR 1982 Report [U6], and some aspects were further evaluated in the UNSCEAR 1988 and 1993 Reports [U3, U4]. The topic is discussed further, with updated information, in Chapter III of this Annex. Many persons are also exposed to enhanced levels of natural radiation at their places of work. Such workers include underground miners, some workers involved in processing of minerals, and aircraft flight crew. Occupational radiation exposures from both man-made and natural sources are considered in Annex E, "*Occupational radiation exposures*".

6. The broad relevance of natural background exposures to the world population makes the evaluations of this Annex particularly pertinent. For most individuals, the natural background exposures are much more significant than the exposures caused by man-made sources. Exceptions that apply to certain individuals are some exposures caused by medical radiation procedures, through mishandling of radiation sources, in accidents allowing radionuclides to be released to the environment, and at some workplaces. In all cases, however, the natural background source forms the baseline upon which all other exposures are added, and it is a common level against which other exposures may be compared.

I. COSMIC RADIATION

7. The earth is continually bombarded by high-energy particles that originate in outer space. These cosmic rays interact with the nuclei of atmospheric constituents, producing a cascade of interactions and secondary reaction products that contribute to cosmic ray exposures that decrease in intensity with depth in the atmosphere, from aircraft altitudes to ground level. The cosmic ray interactions also produce a number of radioactive nuclei known as cosmogenic radionuclides. Best known of these are ^{3}H and ^{14}C. Exposures from cosmic rays and from cosmogenic radionuclides are considered in this Chapter.

A. COSMIC RAYS

8. Galactic cosmic rays incident on the top of the atmosphere consist of a nucleonic component, which in

aggregate accounts for 98% of the total, and electrons, which account for the remaining 2%. The nucleonic component is primarily protons (88%) and alpha particles (11%), with the remainder heavier nuclei [G11]. These primary cosmic particles have an energy spectrum that extends from 10^8 eV to over 10^{20} eV. Below 10^{15} eV the shape of the energy spectrum can be represented by a power function of the form $E^{-2.7}$, where E is expressed in eV. Above that point, known as the knee, the spectrum steepens to a power of −3. The highest energy thus far measured is 3.2 10^{20} eV, which was inferred from ground measurements of the resulting cascade interactions in the atmosphere [O7].

9. It is thought that all but the highest energy cosmic rays that reach earth originate within the earth's own galaxy. The sources and acceleration mechanisms that create cosmic rays are uncertain, but one possibility recently substantiated by measurements from a spacecraft [K16] is that the particles are energized by shock waves that expand from supernova. The particles are confined and continually deflected by the galactic magnetic field. They become isotropic in direction, and the flux is fairly constant in time.

10. Beyond 10^{15} eV, protons may begin to escape the galactic confinement. This leaves relatively greater proportions of heavier nuclei particles in the composition of cosmic rays above this energy level. Protons with energies greater than 10^{19} eV would not be significantly deflected by the intergalactic magnetic field. The fact that protons of such high energy are also observed to be isotropic and not aligned with the plane of the galactic disk suggests that they are probably of extragalactic origin [C7]. Only astrophysical theories can suggest the origins of these ultra-high-energy cosmic rays.

11. Another component of cosmic rays is generated near the surface of the sun by magnetic disturbances. These solar particle events are comprised mostly of protons of energies generally below 100 MeV and only rarely above 10 GeV (10^{10} eV). These particles can produce significant dose rates at high altitudes, but only the most energetic affect dose rates at ground level. Solar particle events can, in addition, disturb the earth's magnetic field in such a way as to change the galactic particle intensity. The events are of short duration, typically a few hours, and highly variable in intensity. They have a negligible impact on long-term doses to the general population.

12. The most significant long-term solar effect is the 11-year cycle in solar activity, which generates a corresponding cycle in total cosmic radiation intensity. The periodic variation in solar activity produces a similar variation in the solar wind. The solar wind is a highly ionized plasma with associated magnetic field, and it is the varying strength of this field that modulates the intensity of galactic cosmic radiation. At times of maximum solar activity the field is at its highest and the galactic cosmic radiation intensity is at its lowest.

13. The magnetic field of the earth partly reduces the intensity of cosmic radiation reaching the top of the atmosphere, the form of the earth's field being such that only particles of higher energies can penetrate at lower geomagnetic latitudes. This produces the geomagnetic latitude effect, with minimum intensities and dose rates at the equator and maximum near the geomagnetic poles.

14. The high-energy particles incident on the atmosphere interact with atoms and molecules in the air and generate a complex set of secondary charged and uncharged particles, including protons, neutrons, pions and lower-Z nuclei. The secondary nucleons in turn generate more nucleons, producing a nucleonic cascade in the atmosphere. Because of their longer mean free path, neutrons dominate the nucleonic component at lower altitudes. As a result of the various interactions, the neutron energy distribution peaks between 50 and 500 MeV; a lower energy peak, around 1 MeV, is produced by nuclear deexcitation (evaporation). Both components are important in dose assessment.

15. The pions generated in nuclear interactions are the main source of the other components of the cosmic radiation field in the atmosphere. The neutral pions decay into high-energy photons, which produce high-energy electrons, which in turn produce photons etc., thus producing the electromagnetic, or photon/electron, cascade. Electrons and positrons dominate the charged particle fluence rate at middle altitudes. The charged pions decay into muons, whose long mean free path in the atmosphere makes them the dominant component of the charged-particle flux at ground level. They are also accompanied by a small flux of collision electrons generated along their path.

16. The changing components of dose rate caused by the secondary cosmic ray constituents in the atmosphere are illustrated in Figure I. At ground level, the muon component is the most important contributor to dose; at aircraft altitudes, neutrons, electrons, positrons, photons, and protons are the most significant components. At higher altitudes, the heavy nuclei component must also be considered.

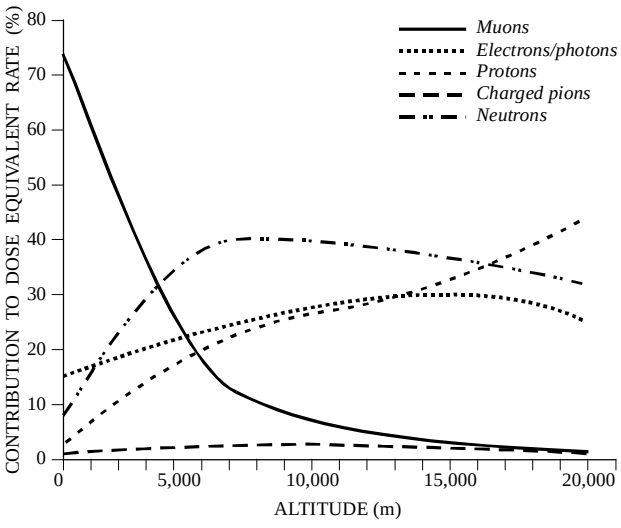

Figure I. Components of the dose equivalent rate from cosmic rays in the atmosphere [O4].

17. The cosmic radiation intensity in the atmosphere has been measured in increasing detail in recent years. A complete mapping of the cosmic radiation field and the determination of exposure conditions and doses throughout the atmosphere as a function of time can be based on these measurements with appropriate interpolation or by the application of reliable radiation transport codes. Codes have been developed for this purpose [O1, W3], and transport codes for accelerator shielding applications have been adapted [K18, P17, R19]. Their adequacy has been, and is currently being, tested against the available measurements.

18. Since the publication of the UNSCEAR 1993 Report [U3], some new information has been added to the database on which the exposure of the general population to cosmic radiation at ground level is based. In particular, both the low- and high-energy peaks in the neutron energy distribution are recognized, and instrumentation has been developed that responds to the extended energy range. This has led to modified estimates of dose from this component of cosmic radiation. There has been substantial progress in recent years in the study of the cosmic radiation fields at aircraft altitudes [E1].

1. Exposure at ground level

19. At ground level, the dominant component of the cosmic-ray field is muons with energies mostly between 1 and 20 GeV. These contribute about 80% of the absorbed dose rate in free air from the directly ionizing radiation; the remainder comes from electrons produced by the muons or present in the electromagnetic cascade. In the early literature, these two components of the charged particle flux were referred to as the "hard" and "soft" components, respectively, because the electrons are much more readily absorbed by any shielding. As altitude increases, the electrons become more important contributors to the dose rate.

20. Many measurements have been made of the altitude profile of the charged-particle and photon ionization and the absorbed dose rate in free air at ground level. A review of this information in the UNSCEAR 1988 Report [U4] indicated that a representative value for this dose rate at sea level is 32 nGy h^{-1}. The geomagnetic latitude effect is about 10%, so that a value of 30 nGy h^{-1} is appropriate for latitudes below 30°. Considering that a large fraction of the world population lives at latitudes below 30° (50% in the northern hemisphere, 85% in the southern hemisphere, and 54% overall), the population-weighted average absorbed dose rate from the directly ionizing and photon components of cosmic radiation at sea level corresponds to 31 nGy h^{-1}, although it is not known to this precision. The dose rate values may be considered as averages over the 11-year solar activity cycle, with the total range of variation about 10%. Since mostly muons are involved, a radiation weighting factor of unity is appropriate [I1], yielding the same values for the effective dose rate, i.e. 31 nSv h^{-1} or 270 μSv a^{-1}.

21. It is much more difficult to estimate the neutron contribution to effective dose rate at sea level. Although available data on neutron fluences and energy distributions are sparse, recent measurements and calculations are beginning to provide clarification. Because earlier instrumentation had a low response to high-energy neutrons, which are an important component of the spectrum, some increases in the estimates of the fluence rate and effective dose rate are being suggested. Measurements [R19, S10] made at the top of the Zugspitze mountain in Germany (altitude 2,963 m, atmospheric depth 718 g cm^{-2}) and associated calculations gave a fluence rate of 0.126 ± 0.01 cm^{-2} s^{-1} [S48]. Attenuation with altitude was described using the function $e^{-0.00721p}$, where p (g cm^{-2}) is the atmospheric depth. From this, a fluence rate at sea level (p = 1,033 g cm^{-2}) of 0.0122 ± 0.001 cm^{-2} s^{-1} can be derived. A value of 0.0133 ± 0.001 $cm^{-2}s^{-1}$ was determined at about sea level for a geomagnetic latitude of 53°N near Braunschweig in Germany [A15] and a value of 0.0123 cm^{-2} s^{-1} at sea level for a geomagnetic latitude of 45°N in Hampton, Virginia, in the United States [G20]. Earlier measurement results were 0.008 cm^{-2} s^{-1} [H16, H17].

22. The effective dose rate (resulting from isotropic incidence) at a fluence rate of 0.013 cm^{-2} s^{-1}, obtained by applying a neutron fluence energy distribution weighting factor of 200 pSv cm^2 (equal to 720 nSv h^{-1} per neutron cm^{-2} s^{-1}), is 9 nSv h^{-1} [S48]. The shape of the neutron energy spectrum at habitable altitudes is considered to be relatively invariant, and therefore the fluence to effective dose (isotropic) conversion coefficient is expected to be generally valid. On this basis, the annual effective dose rate from neutrons at sea level and at 50° latitude is estimated to be 80 μSv a^{-1}.

23. Birattari et al. [B19], using a remmeter with an extended range, reported a value corresponding to 80 μSv a^{-1} (±5%), which is in agreement with the estimate derived in the preceding paragraph. From a series of measurements by Burgkhardt et al. [B18] and Gaborit et al. [G16], the sea level effective dose rate from neutrons was determined to be 60 μSv a^{-1}, but these results are probably underestimates, because the instrumentation lacked response to the high-energy component.

24. Incoming protons that initiate the cosmic ray neutron field are strongly affected by the earth's magnetic field, with the effect that the neutron fluence rate in equatorial regions is less than that in polar regions. Investigators have recognized the importance of the latitude effect, but it has not been carefully quantified by reliable measurements. Florek et al. [F14], quoting results of the Los Alamos LAHET code system calculation, suggest that the equatorial neutron fluence rate at sea level is one fifth the polar rate and that the rate at 50° latitude is 80% of the polar rate. Nakamura et al. [N20], combining measurements made at Tokyo (24°N) with those for higher latitudes [H16, H17], obtained a narrower range for the pole to equator variation, i.e. the equatorial rate about one fourth of the polar rate.

25. An approximate analysis of the latitude effect for cosmic ray neutrons at sea level is presented in Figure II. The normalization points are the measurement results of Birattari et al. [B19] at 50°N (9 nSv h^{-1} = 80 μSv a^{-1}) and of Nakamura et al. [N20] at 24°N (4 nSv h^{-1} = 35 μSv a^{-1}). The maximum value is estimated to be roughly 11 nSv h^{-1} (9 nSv h^{-1} ÷ 0.8). The resultant curve may be used to infer the values for 10° latitude bands to be used in deriving a population-weighted average (Table 1). The world average effective dose rate at sea level from cosmic ray neutrons thus determined is 5.5 nSv h^{-1} or 48 μSv a^{-1}.

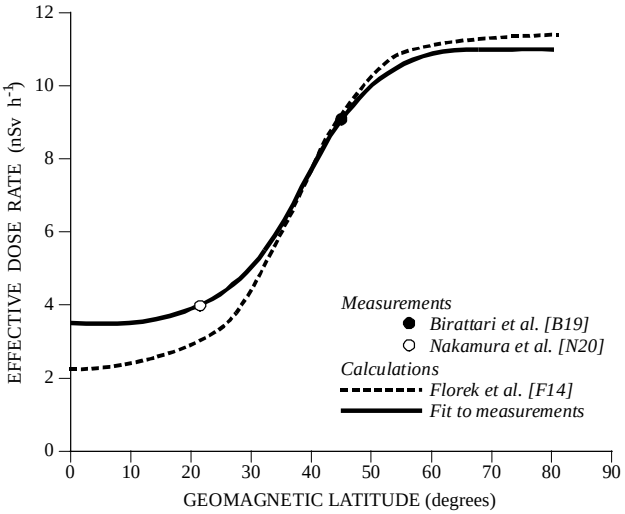

Figure II. Latitude variation in dose rate from cosmic ray neutrons at sea level.

26. For both the directly ionizing and photon component and the neutron component of cosmic rays, there is a substantial altitude effect. Bouville and Lowder [B1] used both measurements and calculations to derive expressions of the altitude dependence of cosmic ray dose rates at habitable locations. These relationships were given in the UNSCEAR 1993 Report [U3] (see also Annex A, "*Dose assessment methodologies*"). Combining these altitude dependence relationships with their analysis of the altitude distribution of the world population, these investigators derived estimates of the population-weighted average dose rates. For the directly ionizing and photon component the population-weighted average dose rate is 1.25 times that at sea level, and for neutrons 2.5 times. Some two thirds of the world population lives in coastal regions, but because dose rates increase with altitude, populations at high altitudes contribute proportionately more to the weighted average. The population-weighted average value corresponds to the dose rate that occurs at 900 m above sea level. The calculations cited by Florek et al. [F14] and the attenuation factor used in paragraph 21 indicate that the effective dose rate from neutrons would increase by a factor of 2.1 between sea level and 900 m elevation, in general agreement with the results of Bouville and Lowder [B1], which were also based on analysis of calculated altitude changes in the dose rate [O3].

27. From estimates derived above, the latitude- and altitude-averaged cosmic ray dose rates may be derived. For the directly ionizing and photon component, the world average effective dose rate is 340 μSv a^{-1} (31 nSv h^{-1} or 270 μSv a^{-1} multiplied by the altitude factor of 1.25); for the neutron component, the average value is 120 μSv a^{-1} (48 μSv a^{-1} multiplied by the altitude factor of 2.5). These results apply to exposures outdoors.

28. The rather limited data on the shielding effect of buildings on cosmic radiation charged particles and photons were summarized in the UNSCEAR 1988 and 1993 Reports [U3, U4]. Observed shielding factors ranged from close to 1 for minimal vertical shielding, e.g. a small wooden house, to 0.4 for lower storeys of substantial concrete buildings. This is consistent with the classical ion chamber observations that defined the "soft" component. These observations imply that a factor of 0.8 would be appropriate after the radiation has passed through a substantial ceiling. In any case, values for particular structures depend on both construction and design, and only broad generalizations can be made. There appears to be no need to change the representative value of the shielding factor, 0.8, used in previous reports.

29. In its previous assessments, the Committee did not apply a shielding factor to the neutron component of cosmic radiation, because of the uncertain balance between attenuation and secondary build-up of neutrons passing through building materials. Although this issue still awaits evaluation, it seems likely that 10%–20% attenuation could be reasonably expected.

30. From the above considerations, the Committee estimates the world average effective dose from the directly ionizing and photon component of cosmic rays to be 280 μSv a^{-1} (applying the indoor shielding factor of 0.8 and assuming indoor occupancy to be 80% of time). The corresponding average value for the neutron component (applying the same adjustment factors) is 100 μSv a^{-1}. The component estimates have been altered slightly from the earlier estimates (300 μSv a^{-1} and 80 μSv a^{-1}) [U3], but the total of 380 μSv a^{-1} remains unchanged. The average annual dose rates for the hemispheres and the world are summarized in Table 2.

31. The global value of the annual collective effective dose is about 2 10^6 man Sv. About one half of this dose is received by the two thirds of the population that lives at altitudes below 0.5 km. The approximately 2% of the population living above 3 km receives a disproportionate 10% of the collective dose. The average annual effective doses from cosmic rays for some high-altitude cities were listed in the UNSCEAR 1993 Report [U3]. Between sea level and 4 km, the neutron contribution to the cosmic radiation effective dose increases from 8% to 35% of the total. Overall, the range of annual average effective doses to the world population is 300–2,000 μSv, with a population-weighted average of 380 μSv.

2. Exposures at aircraft altitudes

32. Aircraft passengers and crew are subject to cosmic radiation exposure rates much higher than the rates at ground level. Total exposure on a given flight depends on the particular path taken through the atmosphere in terms of

altitude (pressure rather than radar altitude) and geomagnetic latitude, as well as on the speed of the aircraft; that is, it depends on the duration of exposure at various altitudes and latitudes. Complicating the situation is the fact that the exposure associated with any flight path may vary with time. There are two possible approaches to dose assessment under these circumstances: (a) area and/or individual monitoring for each flight and (b) determining the radiation fields as a function of time and space and calculating the effective dose for any flight path. Both approaches are being taken, and further measurements and results of calculations are becoming available [E1]

33. Duration of exposure is obviously an important factor in the assessment of doses to passengers and crew. Flight durations for crew members are expressed as the time between leaving the terminal before takeoff and returning after landing. Thus the exposure includes those accrued on the ground and those accrued at all altitudes up to cruising altitude. For flights of more than one hour, the exposure rate at cruising altitude will be the main determinant of dose. The annual number of hours flown by crew members varies from individual to individual and from airline to airline, depending on policy. The range appears to be 300–900 hours per year, with an average of about 500. For the general population, it can be inferred that there are three groups: non-flyers (0 hours), occasional flyers (3–50 hours, with an average of 10), and frequent flyers, i.e. business flyers, couriers, etc. (50–1,200 hours, with an average of 100). The vast majority of the world's population still falls into the first category.

34. Commercial subsonic aircraft generally have cruising altitudes of 7 to 12 km. Although many measurements have been made in aircraft and balloons at these altitudes, there are two major problems in using these data to estimate doses. First, each measurement or set of measurements is carried out for a particular flight path at a particular time, and generalizing such results to other paths and times is not simple. Secondly, most detectors respond to only certain components of the total field, and proper calibration of detector response is generally not simple. In addition, interpreting these data in terms of effective dose requires a knowledge of the overall properties of the complex radiation fields at these altitudes, and this knowledge is as yet incomplete. However, the data can be used as benchmarks to test the ability of existing radiation transport codes to provide reliable information on field properties and effective doses. Moreover, the data obtained by a number of different detector systems on many flights in recent years can be interpreted in terms of the operational quantity ambient dose equivalent, to an estimated accuracy of about 25% [B16, E1, O9, S46, T12].

35. Estimation of doses to passengers and crew are based on the route doses that are obtained from measurements or calculations of the effective dose rate as a function of flight parameters, using, for example, the CARI programme developed by O'Brien and Friedberg [F12, F13, O3] or a computer programme based on measurements and calculations, such as the NASA AIR model [W3] and EPCARD [S47].

36. A working group of the European Commission [E1] reviewed measurements of dose equivalent rates at aircraft altitudes mostly concentrated in the years 1974–1976, when there was minimal solar activity, and 1991, during maximum solar activity. More recent measurement results were presented at the 1998 Dublin Conference [K19]. The results clearly indicate the strong dependence of the dose equivalent rates on altitude, latitude, and the phase of solar activity. The general pattern of measurements is shown in Figure III. The report of the working group [E1] noted that the contributions of the high- and low-LET components are comparable at geomagnetic latitudes of 50° and that the exposure rate throughout the aircraft is approximately constant.

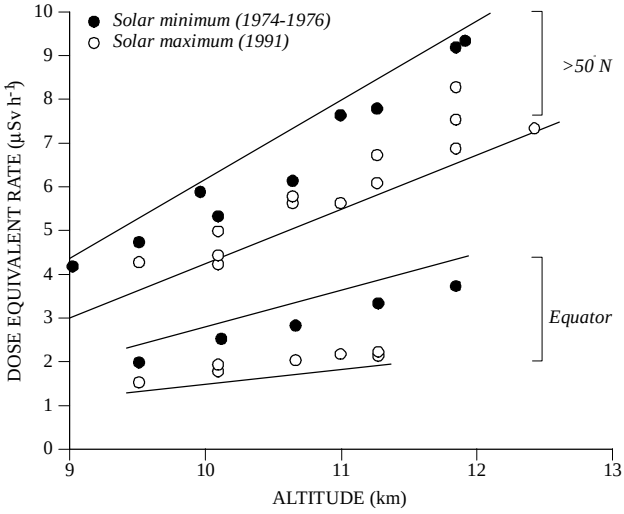

Figure III. Measurement results of cosmic ray exposure rate at aircraft altitudes [E1].

37. The results of recent measurements and recent calculations are broadly consistent. For altitudes of 9–12 km at temperate latitudes, the effective dose rates are in the range 5–8 μSv h^{-1}, such that for a transatlantic flight from Europe to North America, the route dose would be 30–45 μSv. At equatorial latitudes, the dose rates are lower and in the range of 2–4 μSv h^{-1}.

38. A small proportion of passengers and flight crews travel at higher altitudes (~18 km) on supersonic transports. Doses on board those flights are routinely determined from active monitors. Results of this monitoring were summarized in the UNSCEAR 1993 Report [U3]. Effective dose rates of 10–12 μSv h^{-1} were normally found. Recent measurements at these altitudes are in agreement [B24, C29, G21]. One potential problem for high-altitude aircraft is the possibly significant dose contribution from solar particle events. O'Brien et al. [O4] calculated that 13 solar particle events between December 1988 and July 1992 contributed only 2% and 7% of the total cosmic-ray equivalent dose at 11 and 18 km altitude, respectively. However, there is a potential for much more significant events such as the highly energetic event of February 1956. Calculated dose equivalent rates for this event at 20 km are of the order of 1 mSv h^{-1} [A2]. However, no events of this magnitude have taken place since then. It requires both high solar particle flux densities

and high energies (1 GeV) for an event to produce high dose rates at aircraft altitudes, and this is a rare occurrence.

B. COSMOGENIC RADIONUCLIDES

39. The interactions of cosmic-ray particles in the atmosphere produce a number of radionuclides, including ^{3}H, ^{7}Be, ^{14}C, and ^{22}Na. The radioactive half-lives and decay modes of these and other cosmogenic radionuclides with half-lives greater than 1 day are listed in Table 3. Essentially all nuclear species lighter than the target nuclei (primarily nitrogen, oxygen and argon) are produced by high-energy spallation interactions. Production is greatest in the upper stratosphere, but some energetic cosmic-ray neutrons and protons survive in the lower atmosphere, producing cosmogenic radionuclides there as well. Production is not only altitude- but also latitude-dependent and varies as well with the 11-year solar cycle that modulates cosmic-ray penetration through the earth's magnetic field.

40. The calculated global average production rates of cosmogenic radionuclides per unit surface area of the earth and the total annual production are listed in Table 4. The equilibrium global inventory can be derived from the latter value (production rate × 1.44 × half-life). These estimates are somewhat uncertain, as they depend on the validity of the calculational models. Estimates of the environmental distribution of cosmogenic radionuclides can be made based on equilibrium concentrations. The average concentrations in the troposphere are included in Table 4. Since the production, transfer from stratosphere to troposphere, and deposition patterns are latitude- and season-dependent, there may be wide deviations from these average values.

41. Except for ^{3}H, ^{14}C, and ^{22}Na, which are elements with metabolic roles in the human body, the cosmogenic radionuclides contribute little to radiation doses and are mainly of relevance as tracers in the atmosphere and in hydrological systems after deposition. The Committee previously assessed the annual effective doses from cosmogenic radionuclides to be 12 μSv from ^{14}C, 0.15 μSv from ^{22}Na, 0.01 μSv from ^{3}H, and 0.03 μSv from ^{7}Be [U3]. Because of the importance of ^{3}H and ^{14}C from man-made sources of radiation, the environmental and dosimetric aspects of these radionuclides are reviewed in some detail in Annex A, "*Dose assessment methodologies*".

II. TERRESTRIAL RADIATION

42. Naturally occurring radionuclides of terrestrial origin (also called primordial radionuclides) are present in various degrees in all media in the environment, including the human body itself. Only those radionuclides with half-lives comparable to the age of the earth, and their decay products, exist in significant quantities in these materials. Irradiation of the human body from external sources is mainly by gamma radiation from radionuclides in the ^{238}U and ^{232}Th series and from ^{40}K. These radionuclides are also present in the body and irradiate the various organs with alpha and beta particles, as well as gamma rays. Some other terrestrial radionuclides, including those of the ^{235}U series, ^{87}Rb, ^{138}La, ^{147}Sm, and ^{176}Lu, exist in nature but at such low levels that their contributions to the dose in humans are small. Physical data for terrestrial radionuclides are included in Table 3. The external and internal exposures from these radionuclides are evaluated in this Chapter.

A. EXTERNAL EXPOSURES

1. Outdoors

43. External exposures outdoors arise from terrestrial radionuclides present at trace levels in all soils. The specific levels are related to the types of rock from which the soils originate. Higher radiation levels are associated with igneous rocks, such as granite, and lower levels with sedimentary rocks. There are exceptions, however, as some shales and phosphate rocks have relatively high content of radionuclides. There have been many surveys to determine the background levels of radionuclides in soils, which can in turn be related to the absorbed dose rates in air. The latter can easily be measured directly, and these results provide an even more extensive evaluation of the background exposure levels in different countries. All of these spectrometric measurements indicate that the three components of the external radiation field, namely from the gamma-emitting radionuclides in the ^{238}U and ^{232}Th series and ^{40}K, make approximately equal contributions to the externally incident gamma radiation dose to individuals in typical situations both outdoors and indoors.

44. The radionuclides in the uranium and thorium decay chains cannot be assumed to be in radioactive equilibrium. The isotopes ^{238}U and ^{234}U are in approximate equilibrium, as they are separated by two much shorter-lived nuclides, ^{234}Th and ^{234}Pa. The decay process itself may, however, allow some dissociation of the decay radionuclide from the source material, facilitating subsequent environmental transfer. Thus, ^{234}U may be somewhat deficient relative to ^{238}U in soils and enhanced in rivers and the sea. The radionuclide ^{226}Ra in this chain may have slightly different concentrations than ^{238}U, because separation may occur between its parent ^{230}Th and uranium and because radium has greater mobility in the environment. The decay products of ^{226}Ra include the gaseous element radon, which diffuses out of the soil, reducing the exposure rate from the ^{238}U series. The radon radionuclide in this series, ^{222}Rn, has a half-life of only a few days, but it has

two longer-lived decay products, ^{210}Pb and ^{210}Po, which are important in dose evaluations. For the ^{232}Th series, similar considerations apply. The radionuclide ^{228}Ra has a sufficiently long half-life that may allow some separation from its parent, ^{232}Th. The gaseous element of the chain, ^{220}Rn, has a very short half-life and no long-lived decay products.

45. The results of spectrometric analyses of soil samples gathered in different countries are listed in Table 5. These are the *in situ* concentrations. If the concentrations have been reported on a dry basis, representative values of soil moisture of 30% by volume and soil density of 1.6 g cm^{-3} have been assumed. The conversion factor (dry to wet basis) is thus 0.81 (dry weight of 1 cm^3: 1.3 g; wet weight of 1 cm^3: 1.3 g soil + 0.3 g water = 1.6 g; ratio: 1.3 ÷ 1.6 = 0.81).

46. The activity concentration of ^{40}K in soil is an order of magnitude higher than that of ^{238}U or ^{232}Th. In its first assessment of representative concentrations of these radionuclides in soil, in the UNSCEAR 1982 Report [U6], the Committee suggested the values of 370, 25, and 25 Bq kg^{-1} for ^{40}K, ^{238}U and ^{232}Th, respectively. On the basis of the higher levels reported for China and the United States, the Committee revised the values for both ^{238}U and ^{232}Th to 40 Bq kg^{-1} in the UNSCEAR 1993 Report [U3]. A more recently completed country-wide survey in China indicates somewhat lower values [P1, P16]. These and the results for many more countries are included in Table 5. The median values are 400, 35, and 30 Bq kg^{-1}, and the population-weighted values are 420, 33, and 45 Bq kg^{-1} for ^{40}K, ^{238}U, and ^{232}Th, respectively. The results of applying the dose coefficients relating soil concentrations to absorbed dose rate in air [I20, S49] to these values are shown in Table 6. The population-weighted values give an average absorbed dose rate in air outdoors from terrestrial gamma radiation of 60 nGy h^{-1}.

47. Direct measurements of absorbed dose rates in air have been carried out in the last few decades in many countries of the world. The database presented in Table 7 encompasses 70% of the world population. A number of countries have been added since the previous evaluation by the Committee [U3], and several values have been revised based on new information. The population-weighted average is 59 nGy h^{-1}, compared with 57 nGy h^{-1} in the previous assessment [U3]. The average values range from 18 to 93 nGy h^{-1}. A typical range of variability for measured absorbed dose rates in air is from 10 to 200 nGy h^{-1}.

48. Of the values reported in Table 7 of the absorbed dose rate in air outdoors, the lowest are in Cyprus, Iceland, Egypt, the Netherlands, Brunei, and the United Kingdom, all less than 40 nGy h^{-1}, and the highest values are in Australia, Malaysia, and Portugal, all greater than 80 nGy h^{-1}. Exposures inferred from the soil concentration results (Table 5) generally show reasonable agreement with the measured outdoor absorbed dose rate in air (Table 8). A discrepancy of 30% or more may indicate that one or the other survey was not representative for the country. Those countries where there are considerable discrepancies include Luxembourg and Sweden, where the ^{40}K levels in soil are relatively high; Syria and Albania, where all levels of radionuclides in soil are low; and Ireland, where the outdoor measurements are rather low. The surveys were conducted by various means and with different numbers of measurements. The representativeness of each survey cannot be judged. The overall results should be reasonably indicative of the global average.

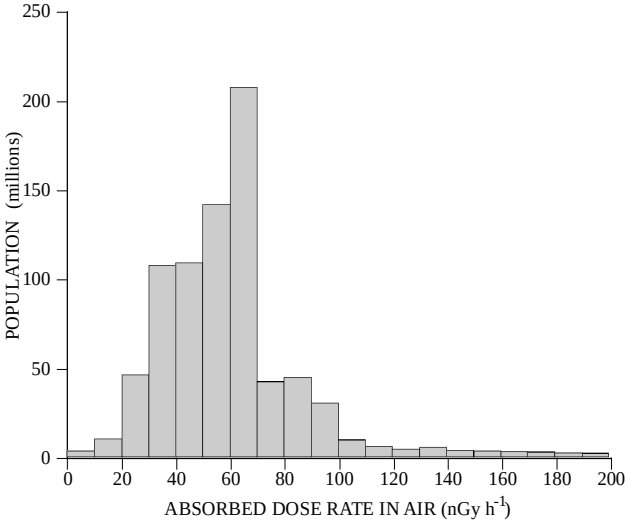

Figure IV. Distribution of population of 25 countries with respect to the outdoor absorbed dose rate in air from terrestrial gamma radiation.

49. A few countries have evaluated the distribution of the population exposed to various ranges of outdoor absorbed dose rates in air. These data, provided in response to the UNSCEAR Survey of Natural Radiation Exposures, are presented in Table 9. The median for the population included (788 million persons in the 25 countries) is in the 50–59 nGy h^{-1} range. A relatively large population group in the Russian Federation is reported to be in the 60–69 nGy h^{-1} range. Decreasing numbers of people are reported to reside in areas with higher levels of outdoor absorbed dose rate in air. The distribution of population according to this sample is presented in Figure IV.

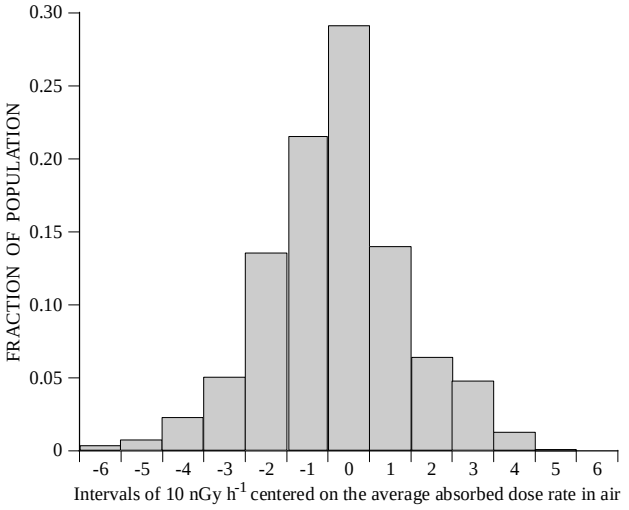

Figure V. Standardized distribution of population with respect to decades about the average absorbed dose rate in air.

50. The total population distribution presented in Figure IV is obtained by combining the data from 25 countries, each with different average outdoor levels of absorbed dose rate in air. The small sample is responsible for the somewhat uneven distribution. The distributions within countries follow a more standard pattern. This is illustrated in Figure V, the data and analysis for which are in Table 10. The distribution of population for each country is centred about a central decade of dose rate indicated as 0. In other words, the distributions are aligned about the central values. Each interval of dose rate represents a decade of dose rate values (e.g. 50–59 nGy h^{-1}). The average distribution is derived from the combined distributions.

51. The standardized distribution is centred about the average level of outdoor dose; 29% of the population is within the 10 nGy h^{-1} decade that encompasses the average value (e.g. is within 50–59 nGy h^{-1} for average outdoor levels anywhere in the range 50–59 nGy h^{-1}). Figure V shows the distribution to ±6 decades of outdoor absorbed dose rate in air. The distribution is relatively normal at levels of dose less than the average, i.e. the population groups are 22% and 14% of the total at the decades of dose from 10 to 20 nGy h^{-1} below the average. The distribution falls more sharply for outdoor levels of dose above the average, i.e. the population groups are 14% and 6% of the total at the next two decades of dose from 10 to 20 nGy h^{-1} above the average. The distribution is approximately log-normal, as shown in Figure VI.

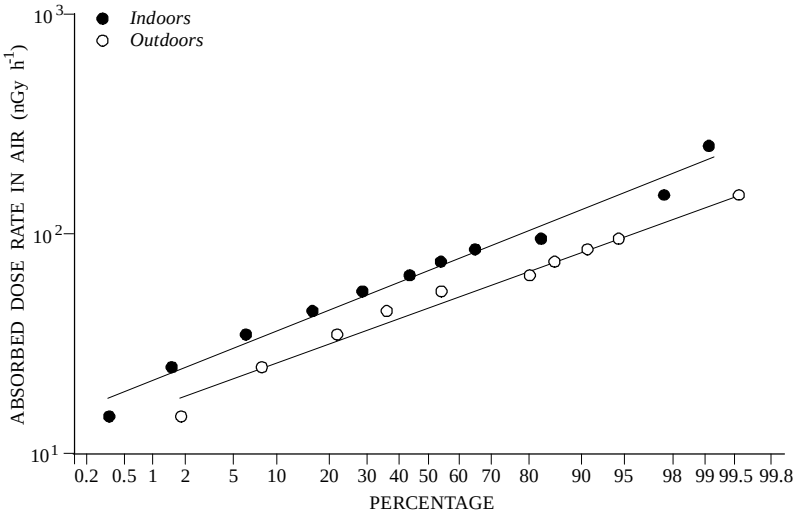

Figure VI. Cumulative distribution of population with respect to outdoor and indoor aborbed dose rate in air from terrestrial gamma radiation.

The data are from independent surveys in different countries (outdoors: Table 9; indoors: Table 12).

52. Although the standardized distribution could be used to indicate the approximate proportions of a population about an average exposure level, it would be important to know whether there are local features of geology that could lead to understandable deviations in the extremes. Extrapolation of the distribution, which is based on just over 10% of the world population, to the entire world population would not be justified, since areas of unusually low or high background levels are probably not well enough represented in the standardized distribution.

53. In addition to variations from place to place, the ambient background gamma dose rate in air at any specific location is not constant in time. It is subject to considerable fluctuation, in particular from the removal of radon progeny in air by rainfall, soil moisture and snow cover. Continuous monitoring records show variations of ±5% from the daily average level in 30-minute measurement intervals [K1, S6]. Washout and rainout of radon progeny from air may result in the short-term enhancement, by 50%–100%, of the gamma-ray dose rate in air. The extent of the elevation depends on rain interval [F2] as well as the rainfall amount. The elevated level lasts for several hours and is followed by a depression of about 5% from the average level, due to shielding from increased soil moisture. If there is no further rainfall, the return to normal occurs in hours or days as the soil saturation disappears. Snow cover depresses the background level by about 1% for each centimetre of snow [F17, H32].

54. There are small areas of markedly high absorbed dose rates in air throughout the world that are associated with thorium-bearing and uranium-bearing minerals in the soil. In those areas, absorbed dose rates in air of several hundred nanograys per hour are not uncommon. The Committee has noted the existence of these areas in all of its previous assessments of natural radiation exposures, and a series of conferences on this topic has helped to bring together the available information [C30, S57, V4, W13].

55. Areas of high natural background are listed in Table 11. There are various causes of these elevated exposure levels. Some result from monazite sand deposits, which have high levels of thorium, including Guarapari in Brazil, Yangiang in China, the states of Kerala and Madras in India, and the Nile delta in Egypt. Some have volcanic soils such as Mineas Gerais in Brazil, Niue Island in the Pacific, and parts of Italy. The central massive in France has granitic and schistic rocks and sands, and an area in the southwest of that country is one

of many associated with uranium minerals in soil. The areas of Ramsar and Mahallat in Iran and are caused by ^{226}Ra deposited from waters flowing from hot springs.

56. It should be noted that exposures in high background areas can vary in time as deposits or beach sands are replenished by springs and tides. Road construction and urbanization of these areas have led to moderate decreases in the background levels [S56, V5].

2. Indoors

57. Indoor exposure to gamma rays, mainly determined by the materials of construction, is inherently greater than outdoor exposure if earth materials have been used; the source geometry changes from half-space to a more surrounding configuration indoors. When the duration of occupancy is taken into account, indoor exposure becomes even more significant. Buildings constructed of wood add little to indoor exposures, which may then be comparable to outdoor exposures.

58. Surveys of absorbed dose rates in air inside dwellings are not as complete as outdoor surveys. The reported values are listed in Table 7. About 45% of the world population is represented in the data that are currently available. The population-weighted average is 84 nGy h^{-1} with national averages ranging from 20 to 200 nGy h^{-1}. The lowest values are in New Zealand, Iceland and the United States, all below 40 nGy h^{-1}, which probably reflects the preponderance of wood-frame houses. The highest values (95–115 nGy h^{-1}) are in Hungary, Malaysia, China, Albania, Portugal, Australia, Italy, Spain, Sweden, and Iran, which must reflect wide use of stone or masonry materials in buildings.

59. The ratios of indoor to outdoor exposure are indicated in Table 7. These are intended not to reflect actual conditions at specific locations but to give a general, relative idea of the broad data gathered in different countries. The indoor and outdoor results may have been derived in separate surveys in locations not closely coordinated. The outdoor levels generally refer to open, undisturbed ground, but sometimes street locations may have been used. The indoor to outdoor ratios range from 0.6 to 2.3, with a population-weighted value of 1.4. Thus indoor exposures (absorbed dose rate in air from terrestrial gamma radiation) are, in general, 40% greater than outdoor exposures. Values less than one are determined only for Thailand, the United States and Iceland, where wood-frame construction is common. High values of the ratio (>2) result from high levels indoors (in Sweden and Hong Kong) relative to outdoors or from low values outdoors (in the Netherlands) relative to indoors.

60. The distributions of populations with respect to indoor exposures have been assessed in several countries. The data are presented in Table 12. The distributions are more or less symmetrical in several countries, e.g. Belgium, Denmark, and Romania. Bulgaria reports a relatively narrow distribution: the population falls mostly in the central three decades of dose rate. By contrast, the distribution in Hungary is very wide, although nearly 50% of the population is in the single decade just above the mean dose rate for the country. The distribution in Italy is also wide and approximately bimodal. The distributions in the Russian Federation, Finland and Lithuania are characterized by separate peaks in the distributions at decades 2 or 3 above the country mean. These various distributions can no doubt be explained by the types of buildings in which the populations live. Data from additional surveys in other countries will be required to indicate a characteristic distribution that might be further generalized.

61. Indoor and outdoor distributions of external exposures are compared in Figure VII. Only countries for which both indoor and outdoor distributions are available (generally the smaller countries of Europe) are included. The comparison shows the shift to higher exposure rates indoors and the somewhat broader distribution of population for the indoor exposure rate. The population-weighted average exposure rates for the countries included in Figure VII are 58 nGy h^{-1} outdoors and 81 nGy h^{-1} indoors, with an indoor/outdoor ratio of 1.4, which is identical to the population-weighted average for the much larger sample of countries in Table 7.

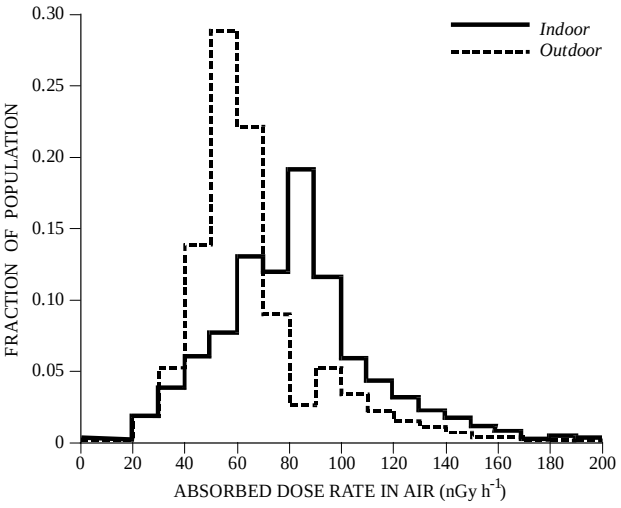

Figure VII. Comparison of indoor and outdoor exposure rates for the total population of nine European countries.

3. Effective dose

62. To estimate annual effective doses, account must be taken of (a) the conversion coefficient from absorbed dose in air to effective dose and (b) the indoor occupancy factor. The average numerical values of those parameters vary with the age of the population and the climate at the location considered. In the UNSCEAR 1993 Report [U3], the Committee used 0.7 Sv Gy^{-1} for the conversion coefficient from absorbed dose in air to effective dose received by adults and 0.8 for the indoor occupancy factor, i.e. the fraction of time spent indoors and outdoors is 0.8 and 0.2, respectively. These values are retained in the present analysis. From the data summarized in this Chapter, the components of the annual effective dose are determined as follows:

Indoors: 84 nGy h^{-1} × 8,760 h × 0.8 × 0.7 Sv Gy^{-1} = 0.41 mSv
Outdoors: 59 nGy h^{-1} × 8,760 h × 0.2 × 0.7 Sv Gy^{-1} = 0.07 mSv

The resulting worldwide average of the annual effective dose is 0.48 mSv, with the results for individual countries being generally within the 0.3–0.6 mSv range. For children and infants, the values are about 10% and 30% higher, in direct proportion to an increase in the value of the conversion coefficient from absorbed dose in air to effective dose.

B. INTERNAL EXPOSURES OTHER THAN RADON

63. Internal exposures arise from the intake of terrestrial radionuclides by inhalation and ingestion. Doses by inhalation result from the presence in air of dust particles containing radionuclides of the ^{238}U and ^{232}Th decay chains. The dominant component of inhalation exposure is the short-lived decay products of radon, which because of their significance are considered separately in Section II.C. Doses by ingestion are mainly due to ^{40}K and to the ^{238}U and ^{232}Th series radionuclides present in foods and drinking water.

64. The dose rate from ^{40}K can be determined directly and accurately from external measurements of its concentration in the body. The analysis of the content of uranium- and thorium-series radionuclides in the body requires more difficult chemical analyses of tissues, and fewer data are available. The analysis of the radionuclide contents of foods and water, along with bioassay data and a knowledge of the metabolic behaviour of the radionuclides, provides an alternative basis for dose estimation. The samples are more readily obtained, and they can reflect widely different locations. With these data, dose estimates for children as well as adults can be derived. The results of both approaches are presented in Section II.B.2.

1. Inhalation

65. Inhalation intake of natural radionuclides other than radon and its decay products makes only a minor contribution to internal exposure. Broadly representative breathing rates are listed in Table 13 for infants (1 year old), children (10 years old), and adults. Results of measurements of the concentrations of uranium- and thorium-series radionuclides in air are listed in Table 14. These radionuclides are present in air because of resuspended soil particles; the decay products of radon are present because of radon gas in air. A dust loading of 50 μg m^{-3} is generally assumed [U6, U7]. With ^{238}U and ^{232}Th concentrations in the soil of 25–50 Bq kg^{-1}, the concentrations in air would be expected to be 1–2 μBq m^{-3}, and this is generally what is observed.

66. It is important to note that the dust loading of air contains substances other than soil, including considerable proportions of organic matter and, especially in wintertime, fly ash from coal burning [K10]. The organic content is deficient in uranium compared to soil, but fly ash contains much higher concentrations of uranium. At coastal locations, concentrations of uranium in sea air may be an order of magnitude lower than in continental or industrialized areas [K11]. Somewhat higher concentrations were measured before 1980, as reported, for example, by Stevenson and Pan [S8]. The subsequent reductions may reflect different fuel supplies.

67. In the UNSCEAR 1993 Report [U3], representative values of the concentrations of terrestrial radionuclides in air were selected. As the database has changed very little, most of these values, as given in Table 14, are still considered valid. The highest concentration is for ^{210}Pb. The concentrations of the other radionuclides are lower by factors of 10, 500, or 1,000 (see Table 14).

2. Ingestion

68. Ingestion intake of natural radionuclides depends on the consumption rates of food and water and on the radionuclide concentrations. The reference food consumption profiles in Table 13 are derived from information on consumption rates adopted by the World Health Organization (WHO) [W1] and food balances compiled by the Food and Agriculture Organization of the United Nations (FAO) [F1]. The values are best interpreted as average values for adults. Consumption rates for children and infants are taken to be two thirds and one third, respectively, of these values, except for milk products, which are consumed in greater amounts by infants and children [C4]. The water intakes are based on reference water balance information from the International Commission on Radiological Protection (ICRP) [I4]. Although the tabulated values are in reasonable agreement with other assessments, there are substantial uncertainties implicit in their mode of derivation. Moreover, there are large deviations from this profile in various parts of the world, e.g. lower milk consumption in Asia and lower leafy vegetable consumption in Africa [W1].

69. Concentrations of naturally occurring radionuclides in foods vary widely because of the differing background levels, climate, and agricultural conditions that prevail. There are also differences in the types of local foods that may be included in the categories such as vegetables, fruits, or fish. In the UNSCEAR 1993 Report [U3], reference values were selected for the concentrations of uranium- and thorium-series radionuclides in foods. Obviously, these values must be derived from the most widely available and representative data possible. The database is summarized in Table 15.

70. It is difficult to select reference values from the wide ranges of concentrations reported for uranium- and thorium-series radionuclides in foods. An example may be made of ^{210}Po, which is present in relatively high concentrations in seafood. The importance of ^{210}Po to dietary intake has been

pointed out for countries such as Japan [Y1], the Marshall Islands [N2], Portugal [C1], and South Africa [H1]. A global review of ^{210}Po in marine food [A3] has suggested that representative concentrations are 2,400 mBq kg^{-1} in fish, 6,000 mBq kg^{-1} in crustaceans and 15,000 mBq kg^{-1} in molluscs. Consumption of fish and shellfish varies widely from country to country and between individuals in a single country. If representative consumption rates are 13 kg a^{-1} of fish and 1 kg a^{-1} each of molluscs and crustaceans, the intake of ^{210}Po with these foods would be 52 Bq a^{-1}. If there are processing or distribution delays for fish products between catch and consumption, the activity intake will be reduced owing to the radioactive decay of ^{210}Po. Statistics quoted by Aarkrog et al. [A3] indicate that 30% of seafood is eaten fresh, 30% frozen, 20% smoked, and 20% canned. For time delays of 0, 1, 2, and 12 months, respectively, the weighted mean time delay is 93 days, slightly less than one physical half-life of ^{210}Po. Application of the correction factor 0.6 suggests an annual intake of 31 Bq in seafood and a weighted concentra-tion of ^{210}Po in fish products of 2,100 mBq kg^{-1}. This result substantiates the reference value of 2,000 mBq kg^{-1} [U3].

71. Estimates of uranium- and thorium-series radionuclides in the total diet are presented in Table 16. They are determined from market-basket evaluations or from duplicate diet samplings. The values derived by multiplying the reference concentrations in foods and water and the intake amounts for adults are shown for comparison. The agreement with presently available data is reasonable.

72. The distributions of the annual intakes in various countries of uranium- and thorium-series radionuclides are shown in Figure VIII. Each point in the Figure represents the average value of the intake for a particular country. If only a range of values has been reported and listed in Table 16, the geometric mean of the extremes of the range has been taken as the representative value. The distributions are approxi-mately log-normal for each radionuclide and span an order of magnitude. Lead-210 and ^{210}Po have the highest concentra-tions and similar distributions, and ^{230}Th and ^{232}Th have the lowest concentrations and also similar distributions. Radium-226 and ^{238}U have intermediate concentrations. Because drinking water is important for the intake of uranium and radium radionuclides, it is necessary to ascertain that this source of ingestion intake is included in the dietary intake estimates.

73. There are a number of circumstances in which the concentrations of natural radionuclides in ingested food and water substantially exceed the reference concentrations or the more typical range of variation. Examples cited in previous UNSCEAR Reports include the Arctic food chain and the high levels of uranium-series radionuclides in well water. Since not all components of the diet are affected and because of common widespread distributions of foods of many different origins over larger regions, the doses to individuals in local populations are not usually so markedly elevated. The circumstances of such exposures should be better described and the data more systematically evaluated.

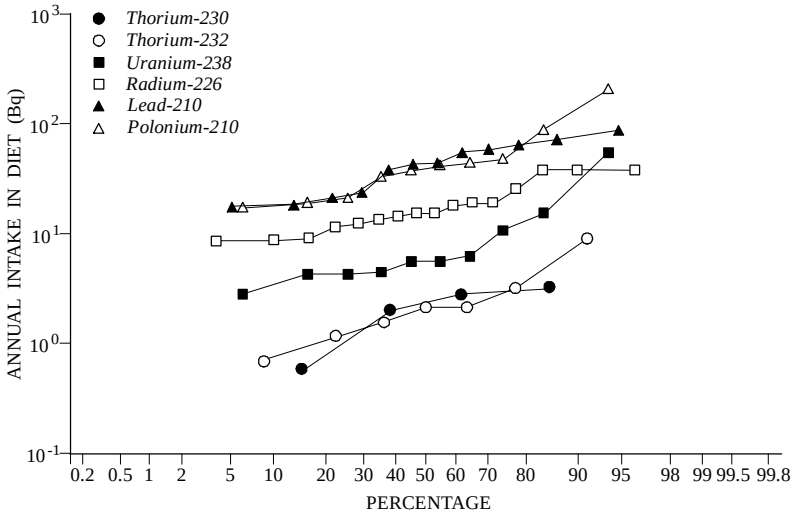

Figure VIII. Ranked distribution of annual intakes of uranium and thorium series radionuclides in diet.
Each point represents the average result of measurements made within a country.

3. Effective dose

74. The evaluation of the internal doses from inhalation is presented in Table 17. Revised dose coefficients taken from ICRP [I9] are used. The age-weighted annual effective dose is 6 μSv from inhalation of uranium- and thorium-series radionuclides in air, which may be compared to the 10 μSv derived in the UNSCEAR 1993 Report [U3].

75. Potassium is more or less uniformly distributed in the body following intake in foods, and its concentration in the body is under homeostatic control. For adults, the body content of potassium is about 0.18%, and for children, about 0.2%. With a natural abundance of 1.17 10^{-4} for ^{40}K [F6], a specific activity of 2.6 10^{8} Bq kg^{-1}, and a rounded dose conversion coefficient of 3 μSv a^{-1} per Bq kg^{-1} [N1], the annual equivalent doses in tissues from ^{40}K in the body are 165 and 185 μSv a^{-1} for adults and children, respectively. The same values are

appropriate for the effective doses, given the more or less uniform distribution of potassium within the body.

76. The evaluation of the internal doses from ingestion of uranium- and thorium-series radionuclides is presented in Table 18. The reference values of concentrations in foods are used with the consumption rates for infants, children, and adults. The age-weighted effective dose assumes a fractional population distribution of 0.05, 0.3, and 0.65, respectively, for infants, children, and adults. Some revisions have been made to the dose coefficients since the UNSCEAR 1993 Report [U3]. The revised values of the dose coefficients [I2] give generally higher estimates of effective dose for these radionuclides. Much of the dose is due to ^{210}Po, for which the gut uptake value recommended by ICRP increased from 0.1 to 0.5. Some of the reference concentrations of ^{210}Pb and ^{210}Po in foods (Table 15) have also been slightly revised. The age-weighted total value is 140 μSv, compared with 52 μSv derived in the UNSCEAR 1993 Report [U3].

77. The total effective dose from inhalation and ingestion of terrestrial radionuclides is 310 μSv, of which 170 μSv is from ^{40}K and 140 μSv is from the long-lived radionuclides in the uranium and thorium series. Essentially the same result is obtained for radionuclide concentrations in body tissues.

78. The Committee reviewed the concentrations of natural radionuclides in tissues in previous assessments, most recently in the UNSCEAR 1988 Report [U4]. Because of the low concentrations in tissues of uranium- and thorium-series radionuclides and variations with age and geographical location, the representative levels remain somewhat uncertain. As additional studies are published only infrequently, this situation is unlikely to change. The database is summarized in Table 19.

79. Uranium is retained in the body primarily in the skeleton. It has been shown that the concentrations are approximately similar in various types of bone (vertebrae, rib, femur) [H23]. Fisenne and Welford [F8] reported that, for residents of New York, concentrations of ^{238}U in vertebrae increased by a factor of 2 over the range 14–73 years and in lungs by a factor of 3 over the same age range. There were no such variations for liver and kidneys. Lianqing and Guiyun [L1] found no variation in concentrations of ^{238}U in bone with age for adult residents of Beijing, but the concentrations in bone of children were up to two times greater than the concentrations in adults. The wide range of concentrations in bone in samples from Beijing (94–2,600 mBq kg^{-1} in dry bone) illustrates the great variations encountered. The generally higher concentrations in Beijing are related to a high intake of ^{238}U in the diet and drinking water [L1].

80. An earlier estimate was that 70% of the body content of ^{238}U was in bone [J9, U6]. This would correspond to 500 mBq in the skeleton (assuming the reference concentration of ^{238}U in bone to be 100 mBq kg^{-1}) and 710 mBq in the body. The average concentration in soft tissues would then be 3 mBq kg^{-1}, although higher concentrations are measured in the lungs and kidneys.

81. Following intake by ingestion and inhalation, thorium is mainly deposited on bone surfaces and retained for long periods. Metabolic modelling assumes that 70% of the body content of thorium is retained in the skeleton [I5]. From the reference concentrations given in Table 19 and assuming the cortical bone mass to be 4 kg and the trabecular bone mass to be 1 kg, it may be estimated that the body burdens are 210 mBq of ^{230}Th and 70 mBq of ^{232}Th.

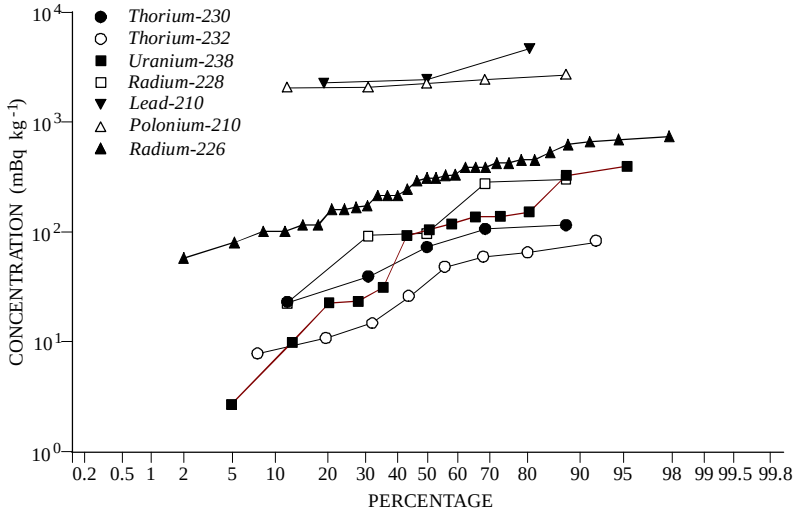

Figure IX. Ranked distribution of the concentrations of uranium and thorium series radionuclides in bone.
Each point represents the average result of measurements made within a country.

82. The distributions of uranium and thorium concentrations in bone in various countries are shown in Figure IX. The values are taken from Table 19. Because the distribution is log-normal within a country, the geometric mean is taken to be the most representative central value. If only a range of values has been reported, the geometric mean of the extremes is plotted in the Figure. The values for individual countries are also distributed approximately log-normally and extend over an order of magnitude, with the variation being caused primarily by differences in intake of the radionuclides in foods

and water. The distributions for ^{238}U and ^{230}Th in bone are similar; somewhat lower concentrations are reported for ^{232}Th. Based on available data, the reference concentrations of uranium and thorium radionuclides given in Table 19 have been revised. These data are limited and must be confirmed as representative for the countries. The concentrations of ^{238}U in soft tissues reported for the former Soviet Union, for example, appear to be abnormally high.

83. Data on ^{226}Ra, ^{210}Pb and ^{210}Po in tissues are also included in Table 19 (the ^{226}Ra data are in summary form). Radium is retained primarily in bone, and concentrations have been measured in many countries. In the UNSCEAR 1977 Report [U7], data from 16 countries were reported, which gave an arithmetic mean value of about 300 mBq kg^{-1} in dry bone. With the fraction of ^{226}Ra in the body distributed in soft tissues taken to be 17% [I3], the average concentration in soft tissues was inferred to be 4.8 mBq kg^{-1}. The population-weighted average of the same data gives somewhat lower values: 230 mBq kg^{-1} in bone and 3.6 mBq kg^{-1} in soft tissues.

84. On the basis of an extended compilation of data from 26 countries, Fisenne [F15] determined the median value of ^{226}Ra in bone from a cumulative population frequency plot to be 170 mBq kg^{-1} in bone. This value was quoted in the UNSCEAR 1982 Report [U6] and accepted as a reference value in the UNSCEAR 1988 and 1993 Reports [U3, U4]. From a further extended series of measurements in 31 countries that include over 60% of the world population, Fisenne [F16] more recently reported a median value of 260 mBq kg^{-1} in bone, inferred from a cumulative frequency plot. The population-weighted averages for these 26 and 31 countries are 230 and 310 mBq kg^{-1}, respectively. Several larger countries with relatively high concentrations in bone have been added to the extended list: Nigeria, 760; Russian Federation, 500; Brazil, 380; and China, 360 mBq kg^{-1}. A higher reference value for ^{226}Ra in dry bone in the range 200–300 mBq kg^{-1} is thus suggested.

85. The only recent data on ^{210}Pb and ^{210}Po concentrations in tissues are those from Japan [T13]. Lead accumulates in bone; by contrast, polonium is distributed mainly to soft tissues. Both would be present in the body in the absence of direct intake from decay of ^{226}Ra, but dietary intake is of most importance in establishing body contents. Early measurements showed the $^{210}Pb/^{210}Po$ concentration ratios to be 0.8 in bone, 0.5 in lungs, and generally 1 in other soft tissues [U7]. Some enhancement of ^{210}Po in liver and kidneys seems substantiated by the data in Table 19. The presence of ^{210}Pb and ^{210}Po in tobacco greatly increases the intake of these radionuclides by smokers. The measured ^{210}Po concentration in the lung parenchyma of smokers is about 3 times that of non-smokers [C32, H35].

86. The annual effective dose from the reference values of uranium–and thorium–series radionuclides in tissue was evaluated in the UNSCEAR 1988 Report [U4]. The estimate was adjusted with revised tissue weighting factors in the UNSCEAR 1993 Report [U3]. The result was 130 μSv. Some changes have been made in the reference concentrations in the present evaluation, and an adjusted value of 120 μSv is obtained. The main contributor to this dose is ^{210}Po. The details of this evaluation are presented in Table 20. This result is in close agreement with the estimate of 110 μSv derived from the dietary consumption of adults and the reference concentrations in foods and water (Table 18).

87. Further data from direct measurements of radionuclides in human tissues would be needed to establish more broadly based estimates of the annual effective dose from internal radionuclides. Studies involving measurements of both radionuclide intake and tissue contents in particular populations would be especially useful to better define the sources and variations in exposures and the magnitudes of the uncertainties in the estimated doses. However, because of the limited number of samples available and therefore the difficulties in determining representative concentrations of natural radionuclides in tissues, it may be necessary to put more reliance on the more widely based dietary intake data for dose estimation purposes.

C. RADON AND DECAY PRODUCTS

88. Radon and its short-lived decay products in the atmosphere are the most important contributors to human exposure from natural sources. While the health risks associated with high radon exposures in underground mines have been known for a long time, relatively little attention was paid to environmental radon exposures until the 1970s, when some scientists began to realize that indoor radon exposures could be quite high, in some cases comparable to the exposures experienced by many underground miners. Since then, the flood of information on radon continues unabated. Many of the more recent papers on the subject have appeared in the proceedings of international conferences at Salzburg (1991), Rimini (1993), Montreal (1995), Prague (1995), Fukuoka (1997), and Athens (1999) [C2, E8, H2, J1, K13, S65], and a valuable synthesis of European research on the subject has recently been published [E2]. All of this information is improving the understanding of the environmental processes that affect radon exposure, but there are still many problems associated with the accurate assessment of exposures and doses to individuals and populations.

89. It is well known that inhalation of the short-lived decay products of ^{222}Rn, and to a lesser extent the decay products of ^{220}Rn (thoron), and their subsequent deposition along the walls of the various airways of the bronchial tree provide the main pathway for radiation exposure of the lungs. This exposure is mostly produced by the alpha particles emitted by several of these radionuclides, although some beta particles and gamma radiation are also emitted. There is general agreement among scientists that it is the alpha particle irradiation of the secretory and basal cells of the upper airways that is responsible for the lung cancer risk seen in miners, although there remains some uncertainty as to exactly which cells are most important for the subsequent induction of lung cancer. It is this situation that is central to the problem of dose assessment. The key

point is that alpha particles emitted into the walls of the airways have a short range, tens of micrometers, and there are large variations in the density of ionizations and excitations along and near the tracks. Thus, the damage to the critical target cells of the respiratory tract depends in a sensitive manner on the source/target geometry. It follows that the dose that is relevant to risk depends critically on those environmental factors that affect the probability that the radon decay products are deposited near the critical target cells after inhalation, as well as on the overall inhalation rate of these decay products. In the following paragraphs, the current concept of radon exposure is described and information on how various environmental factors influence such exposure is summarized, along with available data on exposure levels outdoors and indoors. Absorbed doses to the critical cells and effective doses are then determined by applying the exposure-to-dose conversion factors.

90. The radioactive properties of ^{222}Rn and ^{220}Rn and their respective short-lived decay products are given in Table 3. The various half-lives of the radionuclides are very important in determining the relative contributions of the two series to bronchial dose. The half-life of ^{222}Rn is 3.824 d. It has four short-lived decay products: ^{218}Po (3.05 min), ^{214}Pb (26.8 min), ^{214}Bi (19.9 min), and ^{214}Po (164 μs). Both polonium isotopes are alpha-emitters. The relatively short half-life of ^{220}Rn (55.6 s) means that it does not have much time to travel from its production site to the immediate environment of human beings. The relatively long half-life of one of its decay products, ^{212}Pb (10.6 h), allows this isotope time to deposit on surfaces or migrate away from its source before producing the important alpha-emitter ^{212}Bi (60.6 min). The relative concentrations of the various radionuclides in the two series are also strongly affected by dynamic processes, including the attachment of the decay products to aerosol particles and their subsequent deposition on room surfaces or the ground as well as air movement in general. The fraction of radon progeny in an ultrafine mode (0.5–2 nm), not attached to ambient aerosol particles, is known as the unattached fraction [H5, T16].

91. The evaluation of exposure to radon and the decay products must thus take account of the actual activity concentrations of the various alpha-emitting radionuclides in the two series in the air that is breathed. This consideration, as well as the fact that it is the total alpha particle energy yet to be released by, or following, the decay of inhaled radionuclides that is important in determining dose, has led to the definition of radon exposure rate in terms of potential alpha energy concentration (PAEC) with unit of J m^{-3} or of WL (working level). This quantity can be readily calculated once the activities of the individual radionuclides have been determined from measurement. In most cases, the individual activities are not directly measured, so that the exposure rate must be indirectly determined using assumptions made on concentration ratios, i.e. equilibrium factors, leading to the determination of the equilibrium equivalent concentration. The essential point here is that environmental factors that influence concentration ratios in each of the radioactive series are of great significance for both exposure and dose assessments.

1. Sources of radon

(a) Entry into the atmosphere

92. Radon-222 and ^{220}Rn are the gaseous radioactive products of the decay of the radium isotopes ^{226}Ra and ^{224}Ra, which are present in all terrestrial materials. Some of the atoms of these radon isotopes are released from the solid matrix of the material by recoil when the radium decays. For a radon atom to escape from the mineral grain into the pore space, the decay must occur within the recoil distance of the grain surface. The range of recoil distance for ^{222}Rn is 20–70 nm in common minerals, 100 nm in water, and 63 μm in air [T2]. Radon atoms entering the pore space are then transported by diffusion and advection through this space until they in turn decay or are released into the atmosphere (exhalation). The processes of radon emanation and transport, particularly in the soil, have been reviewed in several classic papers by Tanner [T1, T2]. New studies have focused on the effect of moisture, the dynamics of release or recoil from minerals, radon behaviour in soils as well as on aspects of geology and climate [G22, S50, S59, S60, W9]. Radon generation and transport in porous materials involve solid, liquid, and gas phases in the processes of emanation, diffusion, advection, absorption in the liquid phase, and adsorption in the solid phase. Most aspects of these processes have been characterized individually; however, practical applications require a unified theoretical framework that considers the processes simultaneously [N6, R11].

93. The fraction of radon atoms released into rock or soil pore space from a radium-bearing grain is called the emanation coefficient, the emanation factor or the emanating power. Factors affecting the emanation coefficient were reviewed by Schumann and Gundersen [S50]. Typical emanation coefficients for rocks and soils range from 0.05 to 0.7 [N19]. Grain size and shape are two important factors that control the emanation of radon in soil. They determine how much radium is near enough the grain surface to allow radon to escape into pore spaces. Generally the radon emanation factor is inversely proportional to grain size. The presence of radium in increased concentrations in surface coatings of the grains increases the emanating power relative to that in which radium is uniformly distributed throughout the grains. The sorption or co-precipitation of radionuclides with metal oxides [G18] or organic compounds [G17] in grain coatings is one of the most important processes enhancing the radon emanation coefficient. A study of granitic esker sand showed a high degree of radioactive disequilibrium between ^{226}Ra and ^{238}U, caused by ^{226}Ra adsorbed on the surface of mineral particles [E5]. Microscopic fractures and fissures, called nanopores, and pits or openings caused by previous radioactive decays provide additional pathways for radon release. Particularly in sand-sized and larger grains, nanopores can increase the specific surface area of the grain, enhancing emanation by one or two orders of magnitude.

94. Soil moisture plays an important role in the emanation of radon and its diffusion in soil, for several reasons. Soil moisture, in the form of a thin film of water surrounding soil

grains, directly affects radon emanation by capturing the radon recoils from the solid matrix. These captures increase the likelihood that radon atoms will remain in the pore space instead of crossing the pores and imbedding themselves in adjacent soil grains. Both theoretical estimates [R11] and laboratory tests show that adsorption on soil grains decreases rapidly with increasing water content, becoming insignificant for water contents greater than about 0.3–0.4 of saturation. Decreased adsorption increases the emanation factor at low water contents. Once radon enters the pore space, its partition between the gas and liquid phases depends on the relative volume of water in the pore space and on temperature. The solubility of radon in water decreases with temperature. The partition coefficient of radon between water and gas, the Ostwald coefficient K_T, gives the ratio of concentrations of radon in water and in air [A4, C28, W9]. The value of K_T varies from 0.53 at 0°C to 0.23 at 25°C in water and is typically 0.30 at 15°C. Both partitioning and increased emanation cause the concentration of radon in the air-filled pores to be higher under moist conditions than under dry conditions [A4, W9].

95. The concentration of radon in soil gas, C_{Rn}, in the absence of radon transport is as follows [N19, W9]:

$$C_{Rn} = C_{Ra}\, f\, \rho_s\, \varepsilon^{-1} (1-\varepsilon) (m [K_T - 1] + 1)^{-1} \qquad (1)$$

where C_{Ra} is the concentration of radium in soil (Bq kg^{-1}), f is the emanation factor, ρ_s is the density of the soil grains (2700 kg m^{-3}), ε is the total porosity, including both water and air phases, m is the fraction of the porosity that is water-filled (also called the fraction of saturation), and K_T is the partition coefficient for radon between the water and air phases. For dry soil, m is zero and the last term on the right side of the equation can be omitted. A warm, moist soil (25°C, K_T = 0.23, m = 0.95) with typical soil parameters (C_{Ra} = 30 Bq kg^{-1}, f = 0.2, ε = 0.25) will have a concentration of radon in pore air of 78 kBq m^{-3}, which is 3.7 times higher than for the same soil under cold and dry conditions (0°C, K_T = 0.53, m = 0.05, C_{Rn} = 21 kBq m^{-3}) [W9].

96. Radon concentrations in soil within a few meters of the surface of the ground are clearly important in determining radon rates of entry into pore spaces and subsequently into the atmosphere. They depend on the distribution and concentrations of the parent radium radionuclides in the bedrock and overburden and on the permeability of the soil. Certain generalizations can be made about the radium concentrations in bedrocks of various types, but there are very large ranges within each type. In general, granites have relatively high radium contents, sedimentary and metamorphic rocks intermediate contents, and basalts and most limestones low contents, although there are many striking exceptions to this rule. Soils are similarly variable in their radium content, and generalizations here are even more difficult. This is due in part to the often complex relationship between the bedrock and its overburden, especially in those higher latitude regions that were subject in the past to glaciation. Radium transfers more readily to vegetation than the parent uranium radionuclides, and the emanation from soil organic matter is more effective than from soil minerals. The effective permeabilities of bedrocks and soils are also highly variable, being related to degree of weathering, porosity, moisture content, and the presence of cracks or fissures. This was demonstrated by Schumann and Gundersen [S50] for different soils and climates in the United States. The regional differences are probably caused by climate-controlled differences in soil weathering processes.

97. The key soil-related parameters characterizing radon transport are the radon diffusion coefficient and the soil-air permeability. The diffusion coefficient relates the gradient of the radon concentration in air-filled pores to the flux. It can be determined in many ways, which may cause confusion. The pore diffusion coefficient D_e is also called the "interstitial" or "effective" diffusion coefficient. It relates the gradient of the radon concentration in air-filled pores to the flux density across the air-filled pore area. The "bulk" diffusion coefficient relates the same gradient to the flux density across the geometric (bulk) area. The pore volume is divided into air-filled and water-filled parts. An approximate relationship states that the bulk diffusion coefficient D is equal to εD_e, where ε is the porosity of the soil. Since the radon concentrations in the air-filled and water-filled parts are not the same, the parameter ε must be replaced by the expression $\varepsilon_a + K_T\, \varepsilon_w$, which takes into account the partitioning [A4, N19, R11].

98. Simple models are needed to determine the key parameters of the diffusion coefficient and the soil-air permeability for radon transport calculations. Rogers and Nielson presented a brief review of such expressions [R12]. They also introduced an updated correlation for the effective diffusion coefficient, which was based on more than a thousand diffusion measurements. The experimental pattern of the effective diffusion coefficient D_e as a function of the volume fraction of water saturation is given in Figure X. At low water content, D_e is a little affected by the water content. At high water content, the pores become blocked by water and the diffusion decreases. Typical porosity values for soil materials are 0.01–0.5, with 0.25 representing an average value [U3]. Typical water saturation fractions are 0.1–0.3 for sand and 0.3–0.95 for loam, silty clay, or clay [N7]. The range of D_e in soil is typically 10^{-7}–10^{-5} m^2 s^{-1}. For soil with a fractional water saturation of 0.2 and a porosity of 0.25, the data in Figure X yield an effective diffusion constant of 2 10^{-6} m^2 s^{-1}, which is used as the representative value for soil beneath the reference house (Table 21). In the case of a dry soil (with a total porosity ε of 0.25), the corresponding bulk diffusion coefficient of 5 10^{-7} m^2 s^{-1} is the product of a soil porosity of 0.25 and a D_e of 2 10^{-6} m^2 s^{-1}. This value of the bulk diffusion coefficient corresponds to the representative value given in the UNSCEAR 1988 and 1993 Reports [U3, U4]. If the partitioning has been taken into account, e.g. at a fraction of saturation m of 0.2 (ε = 0.25, ε_a = 0.20 and ε_w = 0.05) and at a temperature of 15°C (K_T = 0.3), the corresponding bulk diffusion coefficient is lower, 4.3 10^{-7} m^2 s^{-1}.

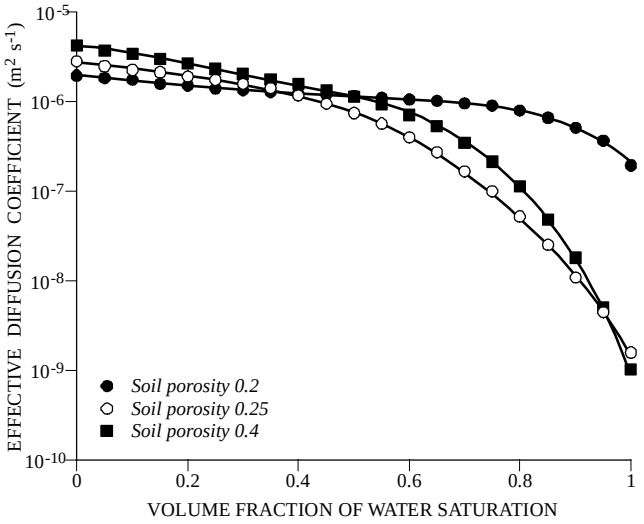

Figure X. Experimental pattern of the effective diffusion coefficient of radon for soil of three different porosities as a function of the fraction of saturation [R12].

99. The main mechanism for the entry of radon into the atmosphere is molecular diffusion. An expression to estimate the diffusive entry rate of radon into the atmosphere was considered in the UNSCEAR 1988 and 1993 Reports [U3, U4]. For a porous mass of homogeneous material semi-infinite in extent, the flux density of radon at the surface of dry soil J_D (Bq m^{-2} s^{-1}) is given by the expression

$$J_D = C_{Ra} \lambda_{Rn} f \rho_s (1-\varepsilon) L \quad (2)$$

where C_{Ra} is the activity concentration of ^{226}Ra in earth material (Bq kg^{-1}), λ_{Rn} is the decay constant of ^{222}Rn ($2.1\ 10^{-6}$ s^{-1}), f is the emanation fraction for earth material, ρ_s is the soil grain density (2700 kg m^{-3}), and ε is the porosity of dry earth material. The diffusion length, L, is equal to $(D_e/\lambda_{Rn})^{½}$. With representative values of these parameters (C_{Ra} = 40 Bq kg^{-1}, f = 0.2, D_e = $2\ 10^{-6}$ m^2 s^{-1}, ε = 0.25), J_D is 0.033 Bq m^{-2} s^{-1}. Equation (2) is valid only for dry soil. The presence of water in soil alters the transport conditions, resulting in a modified equation for J_D. In addition, moisture affects the emanation coefficient and the diffusion coefficient. The estimate of J_D, 0.033 Bq m^{-2} s^{-1}, is in approximate concordance with measured values; however, it is higher than the estimated mean worldwide flux of ^{222}Rn of 0.016 Bq m^{-2} s^{-1} [W8].

100. Although diffusive entry of radon into the outdoor atmosphere usually dominates, there is also some advection caused by wind and changes in barometric pressure. Measurements of exhalation rates of radon from soil show a variability that reflects the variability of radon concentrations in near-surface pore spaces. Concentrations of ^{222}Rn in soil gas vary over many orders of magnitude from place to place and show significant time variations at any given site. Data have shown that there were prominent increases in radon concentrations in outdoor air and in ground water just before the large earthquake at Kobe, Japan, in 1995 [I11, Y3].

101. Under normal circumstances, thoron concentrations in soil gas would be roughly comparable to or perhaps somewhat less than the ^{222}Rn concentrations because of the generally similar production rates in rocks and soils and their similar behaviour in the ground. This has been observed at two locations in New Jersey, United States [H3]. On the other hand, high thoron entry rates from the ground are rarely encountered. Whereas fractures in the ground and/or bedrock allow ^{222}Rn to be pulled to the surface from substantial depths (and volumes), the time frame may be such that most of the thoron present at these depths decays before reaching the surface.

(b) Entry into buildings

102. Knowledge of the factors that influence ^{222}Rn entry rates into structures has considerably improved in recent years as a result of investigations of the processes involved and evaluations of simplified model houses [G1, H4, N16, N19]. In the UNSCEAR 1988 and 1993 Reports [U3, U4], a model masonry building with a volume of 250 m^3, surface area of 450 m^2, and an air exchange rate of 1 h^{-1} was described and calculations carried out to illustrate the effects of the several mechanisms of radon entry, including diffusion and advection from the ground and the building materials, the entry of outdoor air, and ^{222}Rn released from water and natural gas. In the following paragraphs the contributions of these entry mechanisms are reevaluated. The representative soil and house parameters used in the estimation are given in Table 21.

103. Many studies have shown that when high rates of radon entry into buildings are found, advection is usually the main factor [E2, M4]. This advection is driven by the pressure differential between the building shell and the ground around the foundation, produced by the higher temperatures within the shell (the "stack" effect), mechanical ventilation, and to some degree also by wind blowing on the building. The effectiveness of this pressure differential in pulling in radon-laden soil gas through the foundation is critically dependent on the effective permeabilities of both the building foundation and the adjacent earth. Wind can also cause decreases in radon entry concentrations by its flushing effect on radon in soil surrounding the house [R8]. Under certain conditions, atmospheric pressure fluctuations can also represent an important mechanism of radon entry [R13, R14]. Because of differences in the pressure differentials and permeabilities, the advection contribution varies greatly from structure to structure, at least in temperate and cold climates. For non-masonry buildings of similar dimensions in a tropical climate, account must be taken of the usual characteristics and conditions of board floors, calm air, balanced temperatures, and high ventilation (2 h^{-1}). The most important contributions to indoor radon in this case come from outside air and diffusion from the ground, but the total value is not much changed.

104. The effect of anomalous subterranean air flows on indoor radon concentrations has been observed in the United

States in hilly karst terrain [G14, R22] and in Finland on eskers [A5]. Eskers are long and narrow steep-sided ridges formed by glacial streams. In the United States, subterranean networks of cavities and fissures were observed to facilitate advective transport of radon-bearing air. In eskers the coarse sand facilitates underground air flows. In both cases differences between underground and outside air temperatures and the accompanying differences in air density cause subterranean air to move between the upper and lower parts of the area. Wind may also strongly affect the soil air and indoor radon concentrations in these areas. These flows amplify indoor radon levels in winter or summer, depending on the location of the house. Air flows due to thermal differences and seasonal patterns of radon concentrations, which are comparable with the observations described above, have been observed in caves and in mining regions close to the tunnels and air shafts [C27, L17, S39].

105. Modelling studies have helped in understanding the relevance of factors that influence ^{222}Rn entry rates into structures. The modelling results were reviewed by Gadgil [G1]. The main entry route into the model house is the gap between the foundation wall and the floor slab of concrete. The first analytical studies demonstrated the dominant effect of soil permeability; they also showed that once the gap width exceeds 0.5 mm, it no longer markedly increases the entry rate [M32, N8]. In later, mainly numerical model studies, the effect of a subfloor gravel layer, backfill, entry into slab-on-grade houses, and alternative entry routes were modelled [A4, L4, N6, R15, R16]. The gravel layer below the floor slab greatly increases the radon entry rate. Typically, when the ratio of gravel to soil permeability is over 100 and the soil permeability is less than 10^{-9} m^2, the aggregate layer increases the radon entry by a factor of 3–5 [A4, R15].

106. Permeability strongly affects the convective entry of radon into houses. The range of soil air permeability values is very broad, more than eight orders of magnitude, from less than 10^{-16} m^2 for homogeneous clay to more than 10^{-8} m^2 for clean gravel. In a house with a slab-on-grade, the gap between the floor slab and the foundation wall is the most important entry route for radon-bearing soil air. If the slab is otherwise radon-tight, high radon entry rates can only occur by means of advection, and the diffusive entry rate is of minor importance. For moderate permeabilities ($k > 10^{-12}$ m^2), the entry rate is proportional to the permeability and the pressure difference across the gap. The effect of soil permeability, calculated for a model house, is illustrated in Figure XI [R15]. Of great importance is the presence of cracks or fractures of any kind and of any scale in the solid matrix of the material. These magnify the effects of pressure and temperature differentials on the convective transport of radon. Fractures in bedrock formations, cracks in the soil, and similar inhomogeneities in the materials of the foundation of a structure have been identified as direct causes of high radon entry rates into many structures exhibiting high indoor radon concentrations [E2, K14, S3].

107. To estimate the diffusive entry rate from building materials, the flux density from one side of a building element, such as wall and floor, must be known. This is given by the following expression, presented in the UNSCEAR 1988 Report [U4]:

$$J_D = C_{Ra}\,\lambda_{Rn}\,f\,\rho\,L\,\tanh(d/L) \qquad (3)$$

where L is the diffusion length in concrete, given in equation (2), and d is the half-thickness of the slab. The equation is similar to that related to soil [equation (2)], the only difference being the introduction of the hyperbolic term. The parameters of wall materials given in Table 21 and a wall half-thickness of 0.1 m yield an estimate of 0.18 m for the diffusion length in concrete, the corresponding radon flux J_D being 0.0016 Bq m^{-2} s^{-1}. For a floor slab with a half-thickness of 0.05 m and values given in Table 21, the corresponding diffusion length and radon flux are 0.22 m and 0.0008 Bq m^{-2} s^{-1}. Because the diffusion lengths are greater than the half-thickness of the wall and floor, most of the free radon will be exhaled from the structures. Consequently, the thickness of the structure is a dominant factor affecting the radon flux. These flux densities estimated for building materials are about an order of magnitude less than the flux density from the semi-infinitive soil given above.

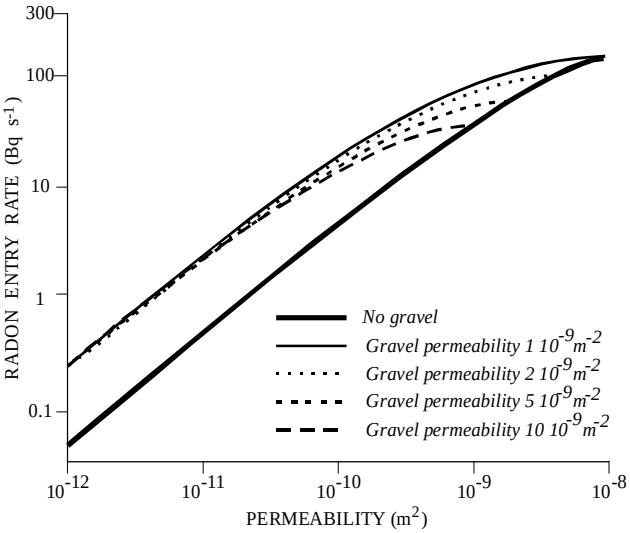

Figure XI. Advective radon entry rates into a typical basement [R15]. *Assumes slab-to-wall gap of 3 mm, deep soil radon concentration of 37 kBq m^{-3}, gravel layer beneath basement slab of thickness of 15 cm and basement pressure of –5 Pa with respect to the atmosphere.*

108. The rate of radon entry from the building elements in the reference house, U (Bq m^{-3} h^{-1}), is given by the expression

$$U = 3.6\,10^3\,S_B\,J_D\,/\,V \qquad (4)$$

where S_B is the surface areas of the walls, J_D is the flux density, and V is the volume of the house (m^3). The surface area of radon-emitting walls in the reference house is estimated to be approximately 450 m^2. The resulting value of U is about 10 Bq m^{-3} h^{-1}. Similarly, the entry rate from

a floor slab with a radon flux density of 0.0008 Bq m^{-2} s^{-1} and a surface area of 100 m^2 is 1.2 Bq m^{-3} h^{-1}. With an air exchange rate of 1 h^{-1}, the corresponding indoor radon concentrations in the reference house attributable to the materials of which the walls and floor slab are built are about 10 and 1 Bq m^{-3}.

109. Radon flux from concrete samples has been observed to vary over two orders of magnitude [S40, U3]. This is caused by differences in the ^{226}Ra content of the material, its porosity, density, and emanation fraction. Generally, radon diffusion from the soil through the concrete has been ignored; however, recent measurements from the United States show that radon diffusion through concrete can be a significant mechanism for radon entry into dwellings [R17]. Basically, this is because the quality of concrete in floor slabs is not as high as that of industrial concretes. The porosities are higher, resulting in higher diffusion constants. The measured effective diffusion coefficients in the extensive study in the United States ranged from 2 10^{-8} to 5 10^{-7} m^2 s^{-1}. These values are consistent with previous values in the literature; the upper limit of the range is extended by a factor of about 5. The geometric mean of 1.4 10^{-7} m^2 s^{-1} is sufficiently high to permit radon diffusion to be a significant mechanism for indoor radon entry under common long-term indoor pressures. The results indicate that the diffusion constant is also related to the porosity. The porosity corresponding to the geometric mean diffusion constant was approximately 0.20. When radon entry into the reference house presented in Table 21 was estimated, 1 10^{-7} m^2 s^{-1} was used for the effective diffusion coefficient of the floor slab. The corresponding estimate of the effective diffusion coefficient, 7 10^{-8} m^2 s^{-1}, presented in the UNSCEAR 1993 Report [U3], has been used for wall materials.

110. For a slab lying on the ground, the radon flux transmitted by the slab can be estimated using an empirical formula, e.g. [U4]. With the parameter values of Table 21, an estimate of 0.0071 Bq m^{-2} s^{-1} is obtained. This estimate is higher by a factor of 6 than the estimate presented in the UNSCEAR 1988 Report [U4], 0.0012 Bq m^{-2} s^{-1}, owing to differences in slab thicknesses, diffusion lengths, and radium concentrations in the soil. With a floor area of 100 m^2 and a flux density, J_T, of 0.007 Bq m^{-2} s^{-1} inserted into equation (4), the radon entry rate for the reference house is estimated to be 10 Bq m^{-3} h^{-1} (Table 22). This result is ten times higher than the radon flux from the slab and is comparable to the flux from walls of the reference house. This yields further an indoor radon concentration of 10 Bq m^{-3} when the air exchange rate is 1 h^{-1}. A comparison estimate is available from Figure XII [A4], which illustrates the entry rate through both the slab and the perimeter gap, including the diffusive and advective components. The parameters used in Figure XII were approximately those used for the reference house, Table 21. The diffusive entry through the slab can be estimated from the entry rate calculated for a permeability of 10^{-13} m^2. In this case, the diffusive entry predominates, and advection through the slab makes a negligible contribution. Figure XII yields an estimate of 0.97 Bq s^{-1}, or 0.0097 Bq m^{-2} s^{-1}, from the slab, which is consistent with the estimate above of 0.007 Bq m^{-2} s^{-1}, when the contribution of the diffusive entry rate from slab material of about 0.002 Bq m^{-2} s^{-1}, included in the estimate in Figure XII, is subtracted. In practice, the cover materials to some extent decrease the radon flux from the floor. In basement houses, diffusion of radon through concrete block walls may be a significant source of indoor radon [L21].

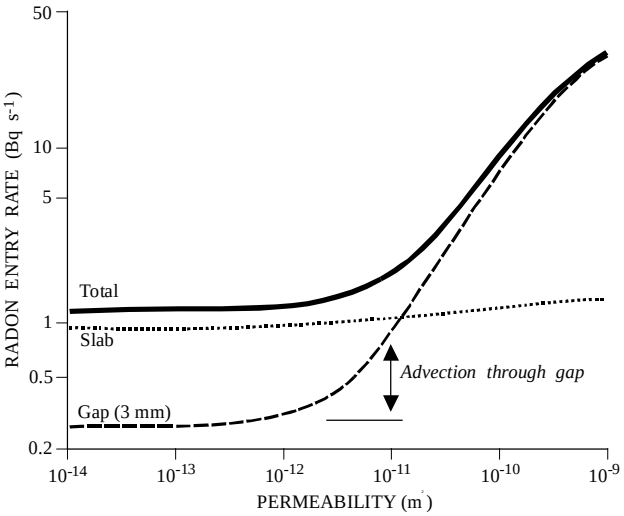

Figure XII. Radon entry rates for model masonry house [A4].

111. The diffusive entry rate through the gap between the floor slab and the foundation wall is next considered. Recent studies [A4, L4, N6, R15] provide improved data from models where diffusive transport is coupled with advective flow. Generally, the increased advective flow through the gaps of the floor slab decreases the relative contribution of diffusion. The upper limit of the diffusive entry rate through the 3 mm perimeter gap can be estimated roughly using the entry rate at the permeability of 10^{-13} m^2 s^{-1} (Figure XII). For this permeability, diffusive entry is the dominant entry mechanism. The estimated volumetric entry rate is 4 Bq m^{-3} h^{-1}. This estimate represents the upper limit for the diffusive entry rate through the gap and has been used as the representative entry rate for the reference masonry house (Table 22).

112. In the published modelling studies, the estimates of the soil-air leakage rate for a house with a basement and a basement wall-floor gap length of 40 m (floor slab of 100 m^2) on soil with permeability of 10^{-10} m^2 are in the range 0.8–2 m^3 h^{-1} [M32, R15]. In a one-floor house with slab-on-grade and a pressure difference of 1 Pa, the corresponding estimate would be 0.2–0.5 m^3 h^{-1}. When a flow rate of 0.2 m^3 h^{-1} and a leakage air concentration of 25,000 Bq m^{-3} are applied to the reference house of Table 21, the advective radon entry rate is 20 Bq m^{-3} h^{-1}. The leakage air concentration is 40% of the deep-soil radon concentration for the reference house. The pressure difference of 1 Pa represents an indoor-outdoor temperature difference of 20°C in a house with slab-on-grade and a natural ventilation system. The advective entry rate of 20 Bq m^{-3} h^{-1} represents a permeability of approximately 2 10^{-11} m^2. This estimate has been used as the representative value for the reference house (Table 22). In the absence of the

gravel layer the permeability would have to be higher by a factor of 2 to yield the same entry rate.

113. The estimates of radon entry into a reference masonry house derived in the preceding paragraphs are summarized in Table 22. Diffusive and advective radon entry each contribute about 40%, and the outdoor air contributes about 20%. The numerical estimates for the various contributions are subject to uncertainties because of the assumptions made. However, the results are roughly consistent with measurements made in actual buildings [H4]. The radon concentration of the reference house is approximately equal to the worldwide average indoor radon concentration. More specific comparisons are made in Table 23 for typical houses in Finland [A1], where the radon concentrations indoors are higher and the air exchange rates are lower than assumed for the reference house. The relative contribution of diffusion sources is lower when the main construction material is wood.

114. Radon dissolved in water may enter indoor air through de-emanation when the water is used. The water supply contribution depends on the concentration of radon in the water used for showering, laundry, etc., and can sometimes be important. The concentrations of radon in water may range over several orders of magnitude, generally being highest in well water, intermediate in ground water, and lowest in surface water. Reference values selected in the UNSCEAR 1993 Report [U3] were 100, 10, and 1 kBq m^{-3} and reference usage was 10%, 30%, and 60%, respectively, for water from the three sources. The ratio of concentrations in air and in water was taken to be 10^{-4} [U3]. This value was also recommended in a national review in the United States of experimental and model study results [N10]. Thus, an average concentration of radon in water of 10 kBq m^{-3} implies a contribution of 1 Bq m^{-3} to radon in air; for an air exchange rate of 1 h^{-1}, the radon entry rate is 1 Bq m^{-3} h^{-1} for the reference house (Table 22).

115. Further evaluation of water as a source of radon for indoor air (public water supplies were measured in 100 major cities of China) confirms these results. The range of radon concentrations in water was 0.04–100 kBq m^{-3}, with an average value of 8 kBq m^{-3} [R9]. Measurements of the air-water concentration ratio did, however, show somewhat higher values, 2–70 10^{-4} on average, in separate studies [R9]. An analysis of all the existing published data giving estimates of the transfer coefficient of radon from water to indoor air derived an average value of 1 10^{-4} [N10].

116. The results of analysing radon entry rates for the reference house suggest at least the relative contributions of the processes involved. The main practical result of such modelling studies has been to identify strategies to mitigate the high radon entry rates through the foundation that are usually the cause of high radon exposures [A11, C25, H19, W4]. These studies have also revealed how complex the situation is with respect to predicting entry rates for individual houses or explaining them when they are measured. Considering all of the factors mentioned above, and especially the design and quality of construction of an individual structure, the factors that determine the entry rate are many, varied, and very site-specific. Successful mitigation strategies, such as identifying and sealing a limited number of entry pathways or effectively ventilating the soil immediately adjacent to the foundation, tend to work because radon entry into many structures can be fairly readily prevented, or at least substantially reduced, by redirecting and re-channelling air transport away from building interiors. Radon concentrations are typically reduced by about 30%. Other techniques aim at reducing the building/ground pressure differential that drives the advection; the radon concentrations are then typically reduced by 80%–99%. Improvements in ventilation systems normally change radon concentrations by less than 50%.

117. The processes that may allow thoron to accumulate in indoor air are difficult to assess. Because of thoron's short half-life, it was once thought that the only mechanisms for significant thoron entry would be infiltration of outdoor air and diffusion from building materials. But recent investigations have shown that entry through the foundation can also be important [L3, S2]. There is an absence of detailed studies in a sufficiently large sample of buildings to make wide generalizations. However, given the comparable concentrations of ^{222}Rn and thoron usually found in outdoor air, soil gas, and building material pore spaces, it is not unexpected that indoor air concentrations of the two gases (ground floor level only) are often roughly comparable.

118. Many of the studies of ^{222}Rn and thoron source terms have dealt with single-family houses, with or without basements and crawl spaces. There is less information on multi-storey buildings, such as apartment houses and office buildings. The expectation that ground sources would be less important for spaces well above the ground has generally been supported by lower measured ^{222}Rn and thoron concentrations in higher storeys [S4], but the ground source contribution depends on air circulation patterns within the building that are both time- and building-dependent.

119. Considerable research has been carried out in recent years to develop methods for defining areas where there is an increased probability of finding buildings with high radon entry rates and indoor air concentrations. A number of models have been developed based on bedrock geology and soil characteristics that have met with only limited success, undoubtedly because of the complications indicated in the preceding paragraphs. Recent efforts in Finland [V3], Japan [F18], Sweden [A14], the United Kingdom [M1], and the United States [G2, P2] have shown that models that incorporate measured radon and radiation levels as well as relevant geological and geophysical parameters are likely to be the most effective.

2. Concentrations in air

(a) Outdoors

120. Concentrations of radon in the outdoor environment are affected not only by the magnitude of the exhalation rates in the general area but also by atmospheric mixing phenomena.

Solar heating during the daytime tends to induce some turbulence, so that radon is more readily transported upwards and away from the ground. At night and in the early morning hours, atmospheric (temperature) inversion conditions are often found, which tend to trap the radon closer to the ground. This means outdoor radon concentrations can vary diurnally by a factor of as much as ten. There are also seasonal variations related to the effects of precipitation or to changes in prevailing winds [B23]. These effects must be taken into account when interpreting the available measurements, many of which are daytime samples.

121. Recent results of radon measurements outdoors tend to confirm the estimates of typical outdoor ^{222}Rn and ^{220}Rn concentrations made in the UNSCEAR 1993 Report of 10 Bq m^{-3} for each [I14]. There is, however, a wide range of long-term average concentrations of ^{222}Rn, from approximately 1 to more than 100 Bq m^{-3}, with the former perhaps typical of isolated small islands or coastal regions and the latter typical of sites with high radon exhalation over large surrounding areas. Although data are sparse for thoron, considerable variability from place to place would be expected because of thoron's short half-life, which means that the effective surface source, about 0.1 km^2 [S4], is much smaller than that for ^{222}Rn, emphasizing the effect of local variations in exhalation rate. Even more important is the fact that thoron's short half-life results in a very steep vertical gradient in its atmospheric concentration at any location. A few measurements show that concentrations a few centimeters above the ground surface and concentrations at a height of 1 m vary by a factor of about 10 [D2, I10, N18]. This gradient would be expected to vary considerably with atmospheric conditions. Thus, pronounced time variations would be expected at any height above the ground at any location. This has obvious implications for estimating thoron exposure outdoors and the outdoor air source term for indoor thoron.

122. Direct measurement of the concentrations of all short-lived decay products of ^{222}Rn and ^{220}Rn are difficult and limited. They are estimated from considerations of equilibrium (or disequilibrium) between these nuclides and their respective decay products. An equilibrium factor F is defined that permits the exposure to be estimated in terms of the potential alpha energy concentration (PAEC) from the measurements of radon gas concentration. This equilibrium factor is defined as the ratio of the actual PAEC to the PAEC that would prevail if all the decay products in each series were in equilibrium with the parent radon. However, it is simpler to evaluate this factor in terms of an equilibrium equivalent radon concentration, C_{eq}, in the following manner:

$$F = C_{eq}/C_{rn}$$

$$C_{eq} = 0.105\ C_1 + 0.515\ C_2 + 0.380\ C_3 \qquad (^{222}\text{Rn series})$$
$$C_{eq} = 0.913\ C_1 + 0.087\ C_2 \qquad (^{220}\text{Rn series})$$

where the symbols C_1, C_2, and C_3 are the activity concentrations of the decay progeny, namely ^{218}Po, ^{214}Pb, and ^{214}Bi, respectively, for the ^{222}Rn series and ^{212}Pb and ^{212}Bi (C_1 and C_2) for the thoron series. The constants are the fractional contributions of each decay product to the total potential alpha energy from the decay of unit activity of the gas. In this way, a measured radon concentration can be converted to an equilibrium equivalent concentration (EEC) directly proportional to PAEC. This provides a measure of exposure in terms of the product of concentration and time. The EEC can be converted to the PAEC, when desired, by the relationships 1 Bq m^{-3} = 5.56 10^{-6} mJ m^{-3} = 0.27 mWL (^{222}Rn) and 1 Bq m^{-3} = 7.6 10^{-5} mJ m^{-3} = 3.64 mWL (thoron).

123. Many measurements have been made of ^{222}Rn and decay product concentrations, allowing estimates to be made of the magnitude of the equilibrium factor to be estimated in terms of both typical values and range. These were discussed in previous reports of the Committee [U3, U4]. More recent extensive measurements in Europe [R1, W10], the United States [W2], Canada [B12], and Japan [H18, K9] indicate typical outdoor ^{222}Rn equilibrium factors of between 0.5 and 0.7. These results suggest that a rounded value of 0.6 may be more appropriate for the outdoor environment than the previous estimate of 0.8. There is, of course, a wide range of values from individual measurements, which is understandable given the many environmental factors that influence the various radionuclide activity ratios, including the exhalation rates and atmospheric stability conditions. The range of the equilibrium factor for outdoor radon is from 0.2 to 1.0, indicating a degree of uncertainty in the application of a typical value to derive equilibrium equivalent concentrations.

124. The equilibrium factor approach is more difficult to apply to estimate thoron decay product exposure because, unlike the ^{222}Rn situation, the concentrations of the gas and the decay products at any particular location, indoors or outdoors, may not be closely related. This is primarily due to the half-lives in the decay series, which produce very different distributions in the atmosphere of the gas and the decay products. A very limited amount of data on thoron decay product concentrations outdoors indicated a typical EEC of the order of 0.1 Bq m^{-3} [S4].

(b) Indoors

125. There is a wealth of data available on indoor ^{222}Rn concentrations, and new information is becoming available on indoor thoron. Substantial compilations of ^{222}Rn results appeared in the UNSCEAR 1988 and 1993 Reports [U3, U4]. These results are supplemented with recent survey data in Table 24. It is sometimes difficult to evaluate the representativeness of results from published reports. New information will be appearing from many countries in Africa, Asia, and South America, partly as a result of the Coordinated Research Programme on Radon in the Environment, sponsored by the International Atomic Energy Agency (IAEA). This will provide a better understanding of how different climates and housing patterns affect radon exposures. At this stage, it does not appear that the survey results have changed markedly from those contained in the UNSCEAR 1993 Report [U3]. In particular, the values of 40 and 30 Bq m^{-3} for the arithmetic and geometric means of the distribution of

worldwide indoor ^{222}Rn concentrations, with a geometric standard deviation of 2.3, still seem reasonable.

126. The geographic (latitudinal) variation in indoor radon concentration was considered in the UNSCEAR 1993 Report [U3]. Although levels at equatorial latitudes should reflect higher ventilation rates because of higher average outdoor temperatures, the general scatter in the results indicated that many other factors are involved. The additional data available from the present survey are included in Figure XIII.

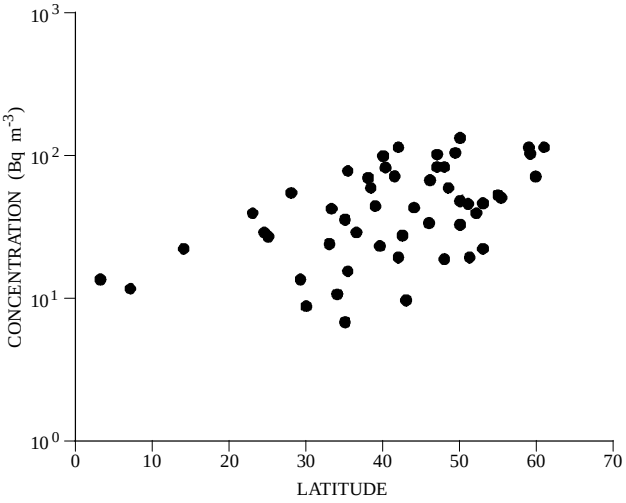

Figure XIII. Average concentrations of radon indoors in various countries in which measurements have been made in relation to latitude.

127. Recent determinations of the equilibrium factor for radon indoors generally confirm the typical value of 0.4 previously assessed by the Committee [U3, U4]. Indoor measurements show a range from 0.1 to 0.9, but most are within 30% of the typical value of 0.4 [H5, R2]. A recent study [H5] in seven North American houses has shown that the equilibrium factor in any building shows a significant variation with time, typically of a few tens of percent. Although the measurement of ^{222}Rn gas concentration may serve as a surrogate for direct measurement of the decay product concentrations in the determination of exposure, it is important to recognize that EECs or PAECs estimated in this manner for particular structures may be in error, frequently by several tens of percent and, rarely, by as much as a factor of 2.

128. Recent research has considerably clarified the situation with respect to indoor thoron and thoron decay product exposures. Several authors, e.g. [M2, N4, S4], have discussed the difficulties and uncertainties in measurements of such exposures and summarized the available data. It is not surprising, based on considerations discussed in previous paragraphs, that the limited data show a wide range of values. This may reflect measurement problems as well as real variations, since various techniques are used, and there has been much less international effort devoted to quality assurance for thoron than for ^{222}Rn. The large uncertainties are also due to the low concentrations usually encountered. The concentrations are highly variable in both space and time and are not closely coupled with the decay product concentrations at a particular location. For example, in a particular room of a structure, the thoron gas concentration varies considerably with distance from the walls and floor because of its short half-life [D1], while the decay products are homogeneously distributed in the room air. Moreover, the decay products were produced partly by thoron present hours earlier, the concentration of which might have been very different. There is, therefore, no surrogate for decay product measurement in the estimation of thoron exposure. This conclusion is supported by recent experimental [M2, M27] and calculational studies [Y2]. Earlier measurement data indicated that a thoron EEC of about 0.3 Bq m^{-3} is fairly typical of indoor atmospheres, although regional averages can be much higher or lower. This value was used in the UNSCEAR 1993 Report [U3].

129. Although measurements of thoron in indoor air are limited, most investigations have reported both the radon and thoron equilibrium equivalent concentrations, so some generalizations from the derived ratios can be made. Based on the physical characteristics of radon and thoron and model entry rates to buildings, ICRP estimated expected concentrations in buildings [I8]. These ranged from 10 to 100 Bq m^{-3} for radon and 2 to 20 Bq m^{-3} for thoron in typical circumstances (3–7 Bq m^{-3} for both radon and thoron in outdoor air; concrete and brick building material; a ventilation rate of 0.7 h^{-1}). In terms of EEC, these concentrations indoors are 2–50 Bq m^{-3} (mean = 15 Bq m^{-3}) for radon and 0.04–2 Bq m^{-3} (mean = 0.5 Bq m^{-3}) for thoron. This corresponds to an expected thoron-radon EEC ratio of 0.03.

130. From regional surveys in the United Kingdom [C26], the ratio of the PAECs of thoron to radon decay products ranged from 0.01 to 30. The highest value was obtained when the ventilation rate in the house was high (2.6 h^{-1}) and the radon concentration was unusually low (2.2 Bq m^{-3}). The distribution of values was approximately log-normal, and most values were between 0.1 and 2. For the wider survey region in the United Kingdom, including areas where high indoor radon concentrations occur, the geometric mean value of the ratio was 0.5. For more typical regions of the country, i.e. excluding the very radon-prone areas, the mean ratio was 0.3 [C26].

131. This ratio can also be expressed in terms of the EEC. The relationship between PAEC and EEC is as follows [I8]: $EEC_{Rn} = 1.81\ 10^8\ C_p$ and $EEC_{Th} = 1.32\ 10^7\ C_p$ for radon (^{222}Rn) and thoron (^{220}Rn) decay products, respectively. The relationships give the EEC with units of Bq m^{-3} when the potential alpha energy concentration, C_p, for either radon or thoron is expressed in J m^{-3}. The thoron-radon EEC ratio is thus lower than the PAEC ratio by a factor of 0.073, so that PAEC ratios of 0.3–0.5 correspond to EEC ratios of 0.02–0.04.

132. Available data on thoron EECs are given in Table 25. These are generally results of a few, short-term measure-

ments and are thus far less representative than the results for ^{222}Rn. Because of the short half-life of ^{220}Rn, the concentration of the gas varied greatly, as mentioned above, with distance from the soil surface or the structural material. Since such measurements cannot easily be standardized, there is little point in presenting data on concentrations of ^{220}Rn alone. The concentration of thoron decay products indoors are highest for wood-frame and mud houses, found particularly in Japan [D1, G12] or with the use of some building materials of volcanic origin, as found in some Italian regions [B25, S62].

133. The previously assumed representative concentrations of thoron EEC were 0.1 Bq m^{-3} outdoors and 0.3 Bq m^{-3} indoors [U3]. These values are at the lower range of values reported in Table 25, most of which were short-term measurements, but are in good agreement with the long-term measurements of Harley and Chittaporn [H36]. The thoron to radon EEC ratio determined in the United Kingdom (0.02) discussed above times the representative radon EEC indoors of 16 Bq m^{-3} (40 Bq m^{-3} × 0.4) would imply a representative value of the concentration of thoron indoors of 0.3 Bq m^{-3}. It thus seems justified to retain the above concentrations of thoron [U3] as representative. Further data are needed on the concentrations of thoron in air in order to provide a reliable estimate of the effective dose from thoron and its decay products.

134. The exposures and consequent doses from radon that are of interest in the assessment of health risks are those integrated over many years. It is well known that there are substantial time variations in the exposure rates from the decay products of both radon nuclides at any location, and each individual spends time at many locations, both indoors and outdoors, where exposure rates can be very different. Much of the published data on indoor ^{222}Rn concentrations is based on time-integrations of days (e.g. using charcoal canisters) to 3–12 months (e.g. using nuclear track detectors). There have been many studies of how short-term measurements can be used to estimate long-term exposures (see, for example, [P2]). One promising development has been the success of models based on outdoor temperature variations (effectively a surrogate for the indoor/outdoor temperature difference) in estimating and tracking the time variations of radon concentration in a few houses [H6]. Local meteoro-logical data can then be used to estimate long-term exposure. In Nordic countries, measurements made indoors in winter, when concentrations are higher because of strong advective air flows from soil, must be adjusted by a correction factor of 0.8 to estimate the annual mean radon concentrations [A12, M28, N12]. In the United Kingdom, correction factors of similar magnitude are needed for short-term measurements in winter and in the opposite direction for such measurements in summer to estimate the average annual concentrations [P11].

135. An important problem in epidemiological radon studies is to determine the long-term average radon levels that existed in the homes of the subjects under investigation. It has been proposed to measure ^{210}Po activities resulting from radon decay on glass surfaces [L2, L19, S52] or in volume traps [O10, S53]. The first technique is based on the deposition of airborne radon decay products onto smooth glass surfaces, followed by their subsequent recoil implantation. The second technique is based on the diffusion of radon throughout the bulk of spongy materials. The radon decay products are directly deposited inside the volume traps, where they remain until they are set free by means of a radiochemical separation procedure. Both of these techniques are promising [F19, M33].

136. The important distinction between “dwelling exposure assessment” and “person exposure assessment” was the subject of a recent experimental study in Austria [S5]. The six-month exposures of 34 individuals were measured with a personal radon meter and estimated from the particular exposure conditions and occupancy times at home, at work, and elsewhere. Results of the two assessments were found to differ by a factor of up to 3, and the possible reasons for these differences were explored. Judicious placement of monitors in the dwelling, for example in a bedroom rather than in the cellar, may reduce the differences between the two assessment approaches [H7].

137. In this Annex, as in previous UNSCEAR reports, typical exposures and exposure conditions are assessed for both indoor and outdoor environments, and doses are estimated from these results and estimated occupancy factors. This assessment is something like the dwelling exposure assessment: the basic assumption is that it reasonably describes typical exposure conditions averaged over large populations. As the population of interest becomes smaller, for example, cases and controls in an epidemiological study, the uncertainties in the assessment of individual and small group long-term exposures must be better understood and quantified.

138. In previous UNSCEAR reports, long-term radon exposures were estimated using indoor and outdoor occupancy factors of 0.8 and 0.2, respectively. These still seem to be reasonable estimates for the global population. However, for smaller population groups and individuals, the factors may be quite different.

3. Effective dose

139. Estimates of absorbed dose to the critical cells of the respiratory tract per unit ^{222}Rn exposure applicable to the general population can be derived from an analysis of information on aerosol size distribution, unattached fraction, breathing rate, fractional deposition in the airways, mucous clearance rate, and location of the target cells in the airways. Such estimates are model-dependent and necessarily subject to all of the uncertainties associated with the input data as well as the assumptions built into the particular calculational model.

140. For both radon-exposed underground miners and those exposed to other carcinogenic aerosols such as cigarette smoke, 75% of lung tumours are found in the branching

airways of the bronchial tree and 15% in the gas exchange region, or parenchyma [S36]. The dosimetry of inhaled radon and decay products is therefore directed to the cells of the bronchial epithelium. The most important variables affecting the alpha dose to the nuclei of these cells are the aerosol size distribution, the unattached fraction, the breathing rate, and the depth in tissue of the target cell nuclei. Considerable research has been conducted to determine quantitative values of the various biological and physical parameters required for lung dosimetry evaluation.

141. Upon decay of ^{222}Rn gas or in recoil from decay of the parent radionuclide, radon decay products are formed as small positive ions or neutral atoms approximately 0.5 nm in diameter. They increase rapidly to 0.5–5 nm as a result of clustering on water or other molecules in the air, depending on local conditions. The ultrafine aerosol mode is called the unattached fraction. Most of these small particles become attached to the local or ambient aerosol, 20–500 nm diameter, and this mode is called the attached fraction. The degree of attachment depends on the ambient aerosol concentration. In dusty, smoky conditions the unattached fraction will be very low, but in a very clean environment, such as prevails with air filtration, the unattached fraction can be much higher. Normally, only ^{218}Po is found in the unattached form with lower concentrations of unattached ^{214}Pb (^{214}Pb/^{218}Po = 1/10).

142. The unattached fraction of radon decay products is expressed as a fraction of the total potential energy (f_{pot}) [I18]. Other expressions have been used involving ratios of decay atoms, so care must be taken in interpreting the data. A central value of the unattached fraction in houses is f_{pot} = 0.05 [H5], but it can vary by a factor of 2 depending on local sources and air filtration.

143. Aerosol characteristics for the indoor environment have been documented by several investigators [B13, H5, N17, R18, T3, T17]. Although the ambient aerosol size in houses is about 100 nm on average, the diameter changes with indoor activities or sources. The use of electric motors, open flames, or electric heaters produces smaller aerosols with diameters of about 50 nm [T3]. Cigarette smoke produces aerosols about 300 nm in diameter [C23]. There is always a distribution of sizes present that can be well represented by a log-normal distribution with a geometric standard deviation of about 2.

144. Breathing rate is an important dosimetric factor because it controls the volume of air brought into the lungs. It can change the dose per unit concentration in air by a factor of about 2, with lower doses derived at lower breathing rates. The breathing rate varies, of course, with the degree of activity of the individual and is not easily measured. The breathing rate of an adult male was estimated to be 0.45 $m^3\ h^{-1}$ resting (8 h per day) and 1.2 $m^3\ h^{-1}$ in light activity (16 h per day) [I4]. The values for the adult female are 20% less resting and 5% less in light activity. The breathing rates were recently re-evaluated [I7], and somewhat lower averages values were derived, namely 22.2 $m^3\ d^{-1}$ for the adult male and 17.7 $m^3\ d^{-1}$ for the adult female.

145. The location of target cell nuclei in the bronchial epithelium has been measured in surgical specimens from over 100 persons of different sex, smoking history, and age [H8]. The average depth of basal and mucous cells implicated in carcinogenesis was 27 and 18 μm, respectively.

146. Deposition of aerosols in the bronchial airways has been investigated with replicate casts from human subjects. The detailed dimensions of the human bronchial airways were first reported by Yeh et al. [Y4] and later verified by Gurman et al. [G15]. Nasal deposition measurements of the unattached fraction is about 15% greater than oral deposition [C24]. Nasal deposition is approximately the same for both cyclic and steady air flow and for all ages. Deposition in the bronchial region occurs by diffusion for aerosol diameters less than 200 nm and by impaction for some particles of greater size. An empirical equation for deposition of aerosols in the upper bronchial airways was derived by Cohen et al. [C5] from measurements using replicate casts. Equations for deposition by impaction have also been derived [C10, G15, R6, R7].

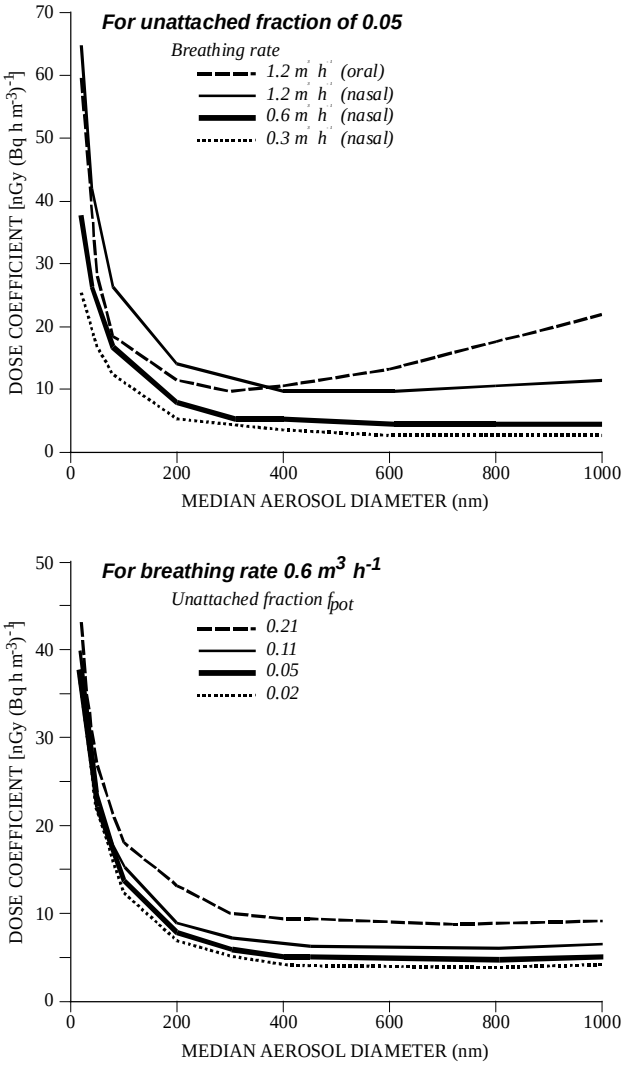

Figure XIV. Absorbed dose in bronchial epithelial cells per unit exposure (EEC) to radon.

147. Utilizing the latest and best available data for the various physical and biological parameters, dosimetrists have

calculated the absorbed dose per unit time-integrated EEC of radon in air. The results of Harley et al. [H8] are illustrated in Figure XIV. Since all particulate exposures contain a distribution of aerosol diameters rather than a unique or monodisperse size, it is more relevant to estimate the dose per unit exposure for a given median diameter and a geometric standard deviation σ_g. The results in Figure XIV are for distributions with $\sigma_g = 2$. The variation in dose with several different breathing rates is shown in the upper portion of Figure XIV for a specific unattached fraction $f_{pot} = 0.05$. The variation of dose with other values of the unattached fraction for a typical breathing rate (0.6 $m^3 h^{-1}$) is shown in the lower portion of Figure XIV. Similar variations are seen for other breathing rates.

148. The dosimetric evaluation of the absorbed dose to basal cells of the bronchial epithelium per unit exposure gives values in the range 5–25 nGy $(Bq\ h\ m^{-3})^{-1}$. The central value is estimated to be 9 nGy $(Bq\ h\ m^{-3})^{-1}$ for average indoor conditions, a breathing rate of 0.6 $m^3\ h^{-1}$, an aerosol median diameter of 100–150 nm and an unattached fraction of 0.05. For an apportioned tissue weighting factor of 0.08 for the bronchial and bronchiolar regions [I7] and the quality factor of 20, the effective dose per unit EEC is 15 nSv $(Bq\ h\ m^{-3})^{-1}$. The dose to the pulmonary region of the lungs is of much less significance.

149. ICRP has developed a more detailed lung model [I7] to calculate the effective dose for exposure to airborne radionuclides. It is, nevertheless, still a simplification of actual respiratory anatomy and physiological behaviour. This model is not yet recommended for radon and its decay products because of the discrepancy in results of risk derived from the model and from epidemiological studies. The difficulties include the measurement and specification of aerosol characteristics, including size and unattached fraction. The model is being used to assess the influence of biological and aerosol parameters and to characterize the uncertainties in estimates of the human lung dose [B11, Z3].

150. The results of major dosimetric studies of the lung dose from deposited ^{222}Rn decay products are compared in Table 26. Differences in the parameter values are evident, but consensus is beginning to emerge on the depth of the target cells, and the characterization of the airways and the deposition measurements based on accurate anatomical representation are greatly improving the dosimetry. With further progress, the derived dosimetric estimates can be expected to converge.

151. Efforts are being made to use measurement techniques based on wire screen penetration theory to simulate the particle collection properties of the nasal and tracheobronchial parts of the respiratory tract [H29]. Several radon progeny samplers for the direct determination of the lung dose have been developed [G19, J11, O2, S54]. Dose coefficients were derived from experimental data using both approaches [H5, H28, R21, S54, W2, Y7]. The results for different indoor and outdoor environments vary from 10 to 50 nSv $(Bq\ h\ m^{-3})^{-1}$. Similar results are obtained in sensitivity studies assessing the influence of biological and aerosol parameters on human lung dose [B11, M25, Z3].

152. As an alternative to a dosimetric approach, ICRP has derived a conversion convention for radon exposures based on the equality of detriments from epidemiological determinations. The nominal mortality probability coefficient for radon for males and females was taken to be 8 10^{-5} per mJ h m^{-3}. This value was determined from occupational studies of miners. Although the exposure conditions in mines are different from those in homes, the differences are compensating, e.g. lower unattached fractions and higher breathing rates in mines than in homes. This coefficient was related to the detriment per unit effective dose, chosen to be 5.6 10^{-5} per mSv for workers and 7.3 10^{-5} per mSv for the public [I1]. The values of the conversion convention are thus 8 10^{-5} ÷ 5.6 10^{-5} = 1.43 mSv $(mJ\ h\ m^{-3})^{-1}$ (5.06 mSv WLM^{-1}) for workers and 8 10^{-5} ÷ 7.3 10^{-5} = 1.10 mSv $(mJ\ h\ m^{-3})^{-1}$ (3.88 mSv WLM^{-1}) for members of the public. The rounded values adopted by ICRP are 1.4 and 1.1 mSv $(mJ\ h\ m^{-3})^{-1}$ (5 and 4 mSv WLM^{-1}) at work and at home, respectively [I6]. The latter value corresponds to 6 nSv $(Bq\ h\ m^{-3})^{-1}$, which is different by a factor of 2.5 from the central value derived using the dosimetric approach. This is not a big discrepancy, considering the complex physical and biological issues involved.

153. The range of dose conversion factors for radon, derived from epidemiological studies [I6] and physical dosimetry, varies from 6 to 15 nSv $(Bq\ h\ m^{-3})^{-1}$. The previous value applied in the UNSCEAR 1993 Report was 9 nSv $(Bq\ h\ m^{-3})^{-1}$. Updated and additional epidemiological studies of 11 underground mining cohorts [C33], performed subsequently to the calculation made in [I6], suggests an increased radon risk per unit exposure. Therefore an increase in the epidemiologically based dose conversion convention is anticipated. The domestic epidemiological radon studies do not yet have sufficient precision to provide numerical risk estimates that could be used in an epidemiological dose conversion convention. Given the present range of values of the dose conversion factor, the established value of 9 nSv $(Bq\ h\ m^{-3})^{-1}$, used in past UNSCEAR calculations [U3, U4], is still considered appropriate for average effective dose calculations. For the representative concentrations of radon selected in Section II.C.2, equilibrium factors of 0.4 indoors and 0.6 outdoors, occupancy, and the dose coefficient as given above, the following annual effective doses are derived:

Indoors: $40\ Bq\ m^{-3} \times 0.4 \times 7{,}000\ h \times 9\ nSv\ (Bq\ h\ m^{-3})^{-1} = 1.0\ mSv$

Outdoors: $10\ Bq\ m^{-3} \times 0.6 \times 1{,}760\ h \times 9\ nSv\ (Bq\ h\ m^{-3})^{-1} = 0.095\ mSv$

154. Less work has been done to derive the dose coefficient for thoron. The values used in the UNSCEAR 1993 Report [U3] were 10 nSv $(Bq\ h\ m^{-3})^{-1}$ for exposures outdoors and 32 nSv $(Bq\ h\ m^{-3})^{-1}$ indoors. Applying the new lung model [I7] results in significantly higher values. There are no epidemiological data for lung cancer risk following thoron exposure from which to derive a conversion convention for

thoron decay products similar to that for radon decay products. The value of 40 nSv (Bq h m^{-3})$^{-1}$ for equilibrium equivalent concentrations of thoron, derived in Annex A, "*Dose assessment methodologies*", seems appropriate for evaluating exposures both indoors and outdoors.

155. The concentration of ^{220}Rn is about 10 Bq m^{-3} outdoors and approximately the same indoors. However, it is not possible to use the concentration of the gas in dose evalua-tion, since the concentration is strongly dependent on the distance from the source. Starting with the assumed represen-tative equilibrium equivalent concentrations, as discussed above, the annual effective dose may be derived as follows:

Indoors: 0.3 Bq m^{-3} (EEC) × 7,000 h × 40 nSv (Bq h m^{-3})$^{-1}$ = 0.084 mSv

Outdoors: 0.1 Bq m^{-3} (EEC) × 1,760 h × 40 nSv (Bq h m^{-3})$^{-1}$ = 0.007 mSv

The average annual effective dose from thoron decay products is thus estimated to be 0.09 mSv.

156. For completeness, the contributions to effective dose from two relatively minor pathways of exposure to radon and thoron can be added, namely dissolution of the gases in blood with distribution throughout the body and the presence of radon in tap water. The dose coefficients for radon and thoron dissolved in blood following inhalation intake are those used in the UNSCEAR 1993 Report [U3]. The dose estimate for radon is

Indoors: 40 Bq m^{-3} × 7,000 h × 0.17 nSv (Bq h m^{-3})$^{-1}$ = 0.048 mSv

Outdoors: 10 Bq m^{-3} × 1,760 h × 0.17 nSv (Bq h m^{-3})$^{-1}$ = 0.003 mSv

For thoron, it is

Indoors: 10 Bq m^{-3} × 7,000 h × 0.11 nSv (Bq h m^{-3})$^{-1}$ = 0.008 mSv

Outdoors: 10 Bq m^{-3} × 1,760 h × 0.11 nSv (Bq h m^{-3})$^{-1}$ = 0.002 mSv.

157. Radon in tap water may lead to exposures from the ingestion of drinking water and from the inhalation of radon released to air when water is used. The concentration of radon in water and the release to air were discussed earlier. Although the calculated result is shown below, this is not a separate contribution to the effective dose, since the radon source from water usage would have been included in the measured indoor radon concentrations. The parameters for the inhalation pathway were presented in paragraph 114: concentration in water of 10 kBq m^{-3}, air-water concentration ratio of 10^{-4}, indoor occupancy of 7,000 hours per year. The inhalation dose coefficient applied is that for the gas. The ingestion of tap water was estimated in the UNSCEAR 1993 Report [U3] to be 100, 75, and 50 l a^{-1} by infants, children, and adults. Assuming the proportion of these groups in the population to be 0.05, 0.3, and 0.65, the weighted estimate of consumption is 60 l a^{-1}. A conservative estimate of the ingestion dose coefficient has recently been evaluated [N10]. The estimated exposures from radon in water are therefore

Inhalation: 10 kBq m^{-3} × 10^{-4} × 7,000 h × 0.4 × × 9 nSv (Bq h m^{-3})$^{-1}$ = 0.025 mSv

Ingestion: 10 kBq m^{-3} × 60 l a^{-1} × 10^{-3} m^3 l^{-1} × × 3.5 nSv Bq^{-1} = 0.002 mSv.

158. The total annual effective dose from radon is thus 1.1 mSv from inhalation of ^{222}Rn and its decay products present in air from all sources, 0.05 from radon gas dissolved in blood, and 0.002 from radon gas in ingested tap water (total = 1.15 mSv). The estimates for thoron are 0.09 from inhalation of ^{220}Rn and its decay products and 0.01 from thoron gas dissolved in blood (total = 0.10 mSv).

159. Considering the range of radon exposures determined from survey data and the generally log-normal distribution of such exposures in particular areas, one would expect to find many large populations around the world (~10^6 individuals) whose average exposures differ from the above-estimated global averages by a factor of more than 2, and many smaller populations (~10^4 individuals) whose average exposures differ by a factor of more than 10. Thus the estimates of the global averages are significant primarily because they define the normal radon and thoron exposures and typical effective doses.

III. ENHANCED EXPOSURES FROM INDUSTRIAL ACTIVITIES

160. There are a number of circumstances in which materials containing natural radionuclides are recovered, processed, used, or brought into position such that radiation exposures result. This human intervention causes extra or enhanced exposures. Although any indoor exposure from building materials surrounding the body would fall in this category, such an exposure is considered a normal component of the natural radiation background. The exposures generally included in the category of enhanced exposures are those arising from the mineral processing industries and from fossil fuel combustion.

161. The Committee generally reviews enhanced exposures in its evaluations of natural radiation sources, as in the latest assessment in the UNSCEAR 1993 Report [U3]. The contribution to the total exposure is usually rather minimal. The UNSCEAR 1982 Report [U6] provided more detailed review of enhanced exposures from natural sources, referring to them at that time as technologically modified exposures. There are also some practices that lead to diminished exposures such as paving roads and using building materials of low radioactive content. These alterations in exposures are usually of less significance than those that cause enhanced exposures.

162. In general, the topic of enhanced exposures is receiving greater attention with several meetings devoted to this subject having recently taken place and several publications issued, e.g. [B26, E6, E7, K20, K21, M34]. Since the Committee has not yet undertaken a wider review of this subject, the reader is referred to the topical publications and proceedings for updated information.

163. In this Chapter, exposures of the general public arising from emissions of naturally occurring radionuclides to the environment from industrial activities are reviewed. Industry uses many different raw materials that contain naturally occurring radioactive materials, sometime abbreviated NORM. These raw materials are mined, transported, and processed for further use. The consequent emissions of radionuclides to air and water lead to the eventual exposure of humans. The main industries are identified below, along with the raw materials and by-products or wastes they generate. The radionuclide content of these raw materials and wastes is summarized in Table 27.

164. ***Phosphate processing***. This industry may be sub-divided into (a) wet processing, (b) thermal processing, and (c) fertilizer production. The primary product is phosphoric acid. In the thermal process, the product may be phosphorus or, using nitric acid, phosphoric acid. Phosphoric acid is used in the manufacture of fertilizers. In the wet phosphate processing industry, phosphogypsum is produced as a by-product. The thermal process (using cokes and silica) produces a slag ($CaSiO_2$) as a waste product.

165. ***Metal ore processing***. Important metal ores are cassiterite or tinstone (tin), tantalite, columbite, fergusonite, koppite, samarskite, and pyrochlore (niobium, iron, manganese, and others). Most of the metals are separated using charcoal or coke. Furnace slag from the processing is often used in cement production. Another by-product, tar coal, is used to produce electrode pitch, creosote oil, carbolineum, soot oil, and road tar mix.

166. ***Uranium mining***. There are several locations that contain the residues of former uranium mining operations, for example, in eastern Germany [B29, E9, R23]. The procedures to deal with the landfills, waste rock and slag piles and the radiological consequences are being evaluated.

167. ***Zircon sands***. Important zirconium minerals are zircon ($ZrSiO_4$) and baddeleyite (ZrO_2). Sorting discriminates these minerals from other heavy minerals or simple silica. The processing involves procedures such as sieving, washing, drying, and grinding. These processes do not produce any specific waste products.

168. ***Titanium pigment production***. Titanium pigments include titanium dioxide (TiO_2) and synthetic rutile. Processing waste products include large quantities of cokes, ore and SiO_2 particles, and filter cake (classified as chemical waste).

169. ***Fossil fuels***. For electric power production the most important fossil fuels are coal, natural gas, and oil. Large amounts of fly ash and bottom ash result from coal combustion. Gypsum is recovered if a desulphurization installation is present.

170. ***Oil and gas extraction***. The large volumes of production water needed for the extraction of oil and gas may contain natural radionuclides, mainly ^{226}Ra and its decay products. Scalings may form as a result of precipitation at the oil/water interface, or radon decay products (especially ^{210}Pb and ^{210}Po) may be deposited in the installations.

171. ***Building materials***. Materials used by the building industry that may be of radiological significance include marl, blast furnace slag, fly ash, Portland clinker, and anhydrate (in the cement industry) and clay (in the ceramics industry). In the cement industry, some silex is produced as a waste product.

172. ***Thorium compounds***. Thorium is used mainly as an additive in other products, such as welding electrodes, gas mantles, and special alloys and is retrieved from monazite, thorite, or thorianite. The activity content of the compounds is present mainly in the primary product, metallic thorium.

173. ***Scrap metal industry***. Scrap metal such as tubing, valves, and heat exchangers from various process industries may contain scales with enhanced levels of natural radionuclides. The particular radionuclides and their concentrations would depend on the origin of the scrap. Since objects from nuclear industries and the uncontrolled releases of radioactive sources may add to this material, which may be recycled, the scrap-metal industry is a source of variable and heterogenous releases of radionuclides into the environment.

174. ***Emissions***. The natural radionuclides present in the raw materials or wastes of these industries are those of the ^{238}U and ^{232}Th series. Releases are mainly to air or water, although landfills after dredging or wastes disposed on land may also provide pathways of exposure.

175. Emissions of radionuclides to air and water from these industries are listed in Table 28. The throughput of ore or raw material is for a typical installation. One of the main radionuclides released to air is ^{222}Rn. It is released by the phosphate and cement industries, gas and oil extraction, gas-fired power production, and, generally, industries that burn natural gas. For example, in the phosphate industry, enhanced radon concentrations between 35 and 780 Bq m^{-3} have been observed, depending on the working area and season [V6]. Important sources of ^{210}Pb to air are the elementary phosphorus and iron and steel production industries. Cement production gives rise to much of the ^{210}Po released. Brick and tile installations may also be of importance because they are so numerous.

176. A special problem is imposed by the storage of uranium-containing minerals in museums [V6]. In a museum in Brussels, where radium- and uranium-containing minerals

from Katanga were stored, concentrations of radon of about 10–15 kBq m^{-3}were found in spite of enhanced ventilation. Besides radon emissions, high levels of gamma radiation were also observed in the vicinity of the storage rooms. In the house of the museum caretaker who lived nearby, gamma levels of 5–6 μSv h^{-1} were found. After shielding of the minerals, the radiation level was reduced to 1–2 μSv h^{-1}.

177. The radionuclides released to the atmosphere by large thermal processes such as those used by elementary phosphorus production, iron and steel production, and the cement industry, are dispersed over great distances. Smaller thermal processes, such as the brick and tile industry, are also sources of airborne releases. For other mineral processing industries, dusty conditions during handling and shipment of ores are the main reason for the releases of radionuclides to air. In those circumstances, the rather coarse particles are generally released mainly to the immediate surroundings of the plant.

178. The largest releases of radionuclides to water are from the phosphate processing, followed by oil and gas production and primary iron and steel production. As an example, two phosphoric acid plants in the Netherlands are responsible for some 90% of all discharges of ^{210}Pb and ^{210}Po to water [L18]. These two plants release about 0.6–0.8 TBq of ^{226}Ra per year [L24], which is comparable to the estimated annual release of ^{226}Ra with process water into the North Sea by the offshore oil production industry in the United Kingdom, Norway, the Netherlands, and Denmark [L25]. Annual releases into rivers of ^{226}Ra and of ^{228}Ra present in diluted brines from single Upper Silesian coal mines may be as high as 20 and 30 GBq, respectively, resulting in locally enhanced concentrations in bottom sediments [L26, S63].

179. The large amounts of gypsum slurry discharged in phosphoric acid production may be released into the sea as is the case in the Netherlands, but industrial wastes are sometimes also stored on land or in large landfills. Radionuclides released to water in, for example, discharges from oil and gas extraction offshore are generally diluted by the large volumes of water involved. Onshore process water is often pumped back into the oilfield. The treatment of production waters before they are released may significantly reduce the radionuclide concentration [L26].

180. Enhanced levels of radionuclides in the environment can come from the processing and use of scrap and recycled metals [B28, L22]. Although in general extensive measures are taken to ensure the continuous quality of the scrap and the new metal that is manufactured from it, enhanced radiation levels are sometimes found. The number of reports on such incidents is growing, partly because of increased awareness of the problem and partly because more measurements are being made. The enhanced exposures may arise from lost radium radiation sources or from naturally occurring radionuclides in pipes with scale containing enhanced concentrations [T15]. Similar problems arise from man-made sources, for example, ^{22}Na, ^{54}Mn [W15], ^{60}Co [C31], ^{137}Cs [B27, J12] or ^{192}Ir, leading to contaminated scrap and recycled metals. The levels vary greatly, and the consequences depend on specific local circumstances.

181. ***Exposures***. Both external and internal exposures may result from naturally occurring radionuclides released by industrial activities. In general, installations are located away from residential areas, and because external radiation levels decrease with distance from the plant, local residents are not significantly exposed. The workers, however, may receive low doses in connection with ore stock piles or waste deposits. Estimated and measured doses are in the range 0.1–300 μSv a^{-1} from direct exposures, with the higher values for locations near mineral-sands-handling industries. The maximum effective doses are summarized in Table 29.

182. Radionuclides dispersed in air may contribute to external irradiation while airborne and after deposition. The contributions to total dose appear to be negligible. Inhalation and ingestion are the pathways that contribute to internal exposure. Inhalation contributes to exposure only in the vicinity of the plant, particularly with mineral-sands-processing plants. Doses depend on distance and could be up to 50 μSv a^{-1} for office workers in a building just outside the plant site [L18].

183. Because most food products consumed by individuals are produced in large agricultural regions, possible dose from ingestion of radionuclides are small. For a typical situation, a small population in the vicinity of an elementary phosphorus plant, the calculated dose would be of the order 100 μSv a^{-1} [L18]. More generally, the estimated doses would be 1–10 μSv a^{-1}. Ingestion doses that could result from discharges of wastes to water are negligible compared to those by the other pathways.

184. In the United Kingdom, the doses from sintering plants of the steel industry to critical groups of the population were calculated to be between about 1.5 and 18 μSv a^{-1}. The highest dose was attributed to a sinter site with relatively low stacks. Inhalation contributed less than 22%, with the main exposure route being the ingestion of food. The annual collective dose calculated for the population (to a distance of 3,000 km) was estimated to be between about 2.9 and 5.5 man Sv [H33].

185. Penfold et al. [P10] made a pilot study of the radiological impact of coal-fired stations in the United Kingdom. Various pathways of exposure were considered. The highest dose rate for a critical group (about 250 μSv a^{-1}) came from the use of fly ash in building materials. Other pathways caused dose rates for critical groups between 0.07 and 55 μSv a^{-1}.

186. The radiation exposure of critical groups of the population surrounding a site with a wood-chip-burning oven was determined in a Swedish study [H34]. The maximum individual dose rate was found to be 2.4 nSv a^{-1}.

187. Annual per caput effective doses from process industries documented in the UNSCEAR 1993 Report ranged from 1 nSv to 20 μSv and for critical groups up to about 1 mSv. Those mentioned above and other more recent data are for very specific situations or critical groups. On the whole, however, they are in agreement with the earlier estimates, and they support the conclusions of the UNSCEAR 1993 Report [U3].

188. ***Summary***. The industrial activities enhancing exposure from natural sources involve large volumes of raw materials containing natural radionuclides. Discharges from industrial plants to air and water and the use of by-products and waste materials may be the main contributors to enhanced exposure of the general public. For typical industries and releases, exposures occur primarily in close proximity to the plants. A complete review is made difficult by the diversity of industries involved and the local circumstances associated with the exposures. Estimated maximum exposures are greatest for phosphoric acid production and the mineral-sands-processing industries. Although exposure rates of the order of 100 μSv a^{-1} could be received by a few local residents, levels of 1–10 μSv a^{-1} would be more common. These exposure rates constitute a negligible component of the total annual effective dose from all natural sources of radiation.

IV. WORLDWIDE AVERAGE EXPOSURE FROM NATURAL SOURCES

189. The components of exposure caused by natural radiation sources have been reassessed in this Annex based on new information and data from measurements and on further analysis of the processes involved. These exposure components can now be added to provide an estimate of the total average exposure. It must first be stated that the average exposure probably does not pertain to any one individual, since there are wide distributions of exposures from each source and the exposures combine in various ways at each location, depending on the specific concentrations of radionuclides in the environment and in the body, the latitude and altitude of the location, and many other factors.

190. In a few countries the proportion of the population at various levels of total exposures has been assessed. These data are included in Table 30, and the combined distribution is shown in Figure XV. The average annual exposure for this distribution is 2.0 mSv. The distribution rises in a few dose intervals to the peak exposure and then tails off to decreasing population at doses 2 to 3 times the average. To smooth the distribution somewhat, most exposure intervals have been subdivided. The general shape of the distribution is probably fairly relevant. Although populations living in areas of high background exposures are not well represented in this particular distribution, they would not be expected to be a prominent feature, in part because not all components of their exposure are enhanced at the same time and because there is a relatively small proportion of the population of most countries with significantly elevated exposures.

191. Average worldwide exposure determined by adding the various components is summarized in Table 31. The changes from the earlier assessment of the Committee [U3] are also indicated. There are only rather minor changes for the exposure components. The worldwide average annual exposure to natural radiation sources remains 2.4 mSv. Neither the magnitude nor the precision of this estimate should be overemphasized. As indicated in Figure XV, based on the sample population of Table 30, a broad distribution of exposures would be expected in any large population.

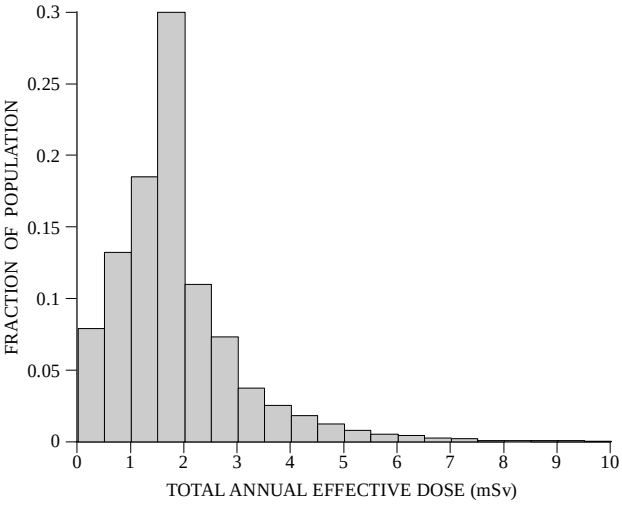

Figure XV. Distribution of population of fifteen countries with respect to total annual effective dose.

192. The normal ranges of exposures to the various components of natural radiation are indicated in Table 31. This accounts for common variations in exposures but excludes those individuals at the extreme ends of the distributions. On this basis, worldwide annual exposures to natural radiation sources would generally be expected to be in the range 1–10 mSv, with 2.4 mSv being the present estimate of the central value.

CONCLUSIONS

193. Since exposures to natural radiation sources are more significant for the world's population than most exposures to man-made sources, the natural background baseline warrants evaluation in some detail. Efforts should continue to broaden the database used for determining both representative values and extremes in exposures and to improve dosimetric procedures.

194. Because of the wide variations in natural background exposures even within relatively small regions, more efforts will be required to determine the detailed distributions of populations within dose intervals for the various components of exposure. Initial, still limited evaluations of distributions of external exposures outdoors and indoors and of the total exposure have been presented in this Annex. These evaluations seem to reveal patterns that would be expected to be generally valid for other countries and for the world population as a whole. The analysis of distributions will provide an improved basis for deriving worldwide average exposures and their normal and extreme variations.

195. The main uncertainties in the assessment of dose from natural radiation sources arise less from the limited number of measurements than from the complications of the dosimetric considerations. The situation with respect to radon decay products is well known, but similar problems exist for cosmic radiation and ingested radionuclides. For cosmic radiation, more information is needed on exposures to neutrons at all altitudes and latitudes, especially high-energy neutrons and high-Z nuclei at aircraft altitudes, along with critical data or improved models to allow a reasonable estimation of effective doses from these components of the radiation field. For ingested radionuclides, good dosimetric models are available, but the problem is to estimate representative intake amounts of the radionuclides and associate them with relatively fewer determinations of concentrations in tissues of the body.

196. There are many circumstances in which individuals receive enhanced exposures to natural radiation. Living inside buildings is considered normal in this regard, and flying in airplanes usually involves an insignificant proportion of most people’s time. In the past, the Committee has reviewed the exposures caused from the release of natural radionuclides in mineral processing industries, the use of phosphate fertilizers, and the combustion of fossil fuels. These enhanced exposures are usually quite insignificant compared with the normal background exposure from natural sources. This conclusion is still valid, based on a brief review of new information in this Annex.

197. The evaluations in this Annex of exposures from natural radiation sources indicate that the average annual effective dose to the world population is approximately 2.4 mSv, which is the same as the previous estimate of the Committee [U3]. The value of the estimated average exposure should not be taken to be too precise, since broad averaging is involved. For individuals, annual exposures ranging from 1 mSv to two or three times the world average are frequently encountered. It is estimated that about 65% of individuals have exposures between 1 and 3 mSv, about 25% of the population have exposures less than 1 mSv, and 10% have exposures greater than 3 mSv. Although the database continues to expand and characterization of the distributions of populations with respect to the various components of natural background radiation is being improved, the generally assessed exposure levels to which the broad spectrum of the world population is exposed seem reasonably well substantiated.

Table 1
Latitude distribution of cosmic ray dose rates outdoors at sea level

Latitude (degrees)	*Population in latitude band (%)*		*Effective dose rate (nSv h^{-1})*	
	Northern hemisphere	*Southern hemisphere*	*Directly ionizing component* [a]	*Neutron component* [b]
80–90	0	0	32	11
70–80	0	0	32	11
60–70	0.4	0	32	10.9
50–60	13.7	0.5	32	10.0
40–50	15.5	0.9	32	7.8
30–40	20.4	13.0	32	5.3
20–30	32.7	14.9	30	4.0
10–20	11.0	16.7	30	3.7
0–10	6.3	54.0	30	3.6
Total	100	100		
Population-weighted average				
Northern hemisphere			31.0	5.6
Southern hemisphere			30.3	4.0
World [c]			30.9	5.5

a Average measurement results.
b From fit to measurements of Figure II.
c Population distribution: northern hemisphere 0.89; southern hemisphere 0.11.

Table 2
Population-weighted average cosmic ray dose rates

Conditions	*Effective dose rate (μSv a^{-1})*						
	Directly ionizing component			*Neutron component*			*Total*
	North	*South*	*World*	*North*	*South*	*World*	*World*
Outdoors, at sea level	272	265	270	49	35	48	320
Outdoors, altitude adjusted [a]	339	332	340	124	87	120	460
Altitude, shielding, occupancy adjusted [b]	285	279	280	104	73	100	380

a Altitude-weighting factors applied to sea level values: directly ionizing component 1.25; neutron component 2.5.
b Building shielding factor 0.8; indoor occupancy factor 0.8.

Table 3
Physical data for radionuclides of natural origin
[F6]

Element	*Isotope*	*Half-life*	*Decay mode*
Cosmogenic radionuclides			
Hydrogen	^{3}H	12.33 a	beta (100%)
Beryllium	^{7}Be	53.29 d	EC [a] (100%)
	^{10}Be	$1.51\ 10^{6}$ a	beta (100%)
Carbon	^{14}C	5730 a	beta (100%)
Sodium	^{22}Na	2.602 a	EC (100%)
Aluminium	^{26}Al	$7.4\ 10^{5}$ a	EC (100%)
Silicon	^{32}Si	172 a	beta (100%)
Phosphorus	^{32}P	14.26 d	beta (100%)
	^{33}P	25.34 d	beta (100%)
Sulphur	^{35}S	87.51 d	beta (100%)
Chlorine	^{36}Cl	$3.01\ 10^{5}$ a	EC(1.9%), beta (98.1%)
Argon	^{37}Ar	35.04 d	EC (100%)
	^{39}Ar	269 a	beta (100%)
Krypton	^{81}Kr	$2.29\ 10^{5}$ a	EC (100%)
Terrestrial radionuclides			
Potassium	^{40}K	$1.28\ 10^{9}$ a	beta (89.3%), EC (10.7%)
Rubidium	^{87}Rb	$4.75\ 10^{10}$ a	beta (100%)
Lanthanum	^{138}La	$1.05\ 10^{11}$ a	beta (33.6%), EC (66.4%)
Samarium	^{147}Sm	$1.06\ 10^{11}$ a	alpha (100%)
Lutecium	^{176}Lu	$3.73\ 10^{10}$ a	beta (100%)
^{238}U series:			
Uranium	^{238}U	$4.47\ 10^{9}$ a	alpha (100%)
Thorium	^{234}Th	24.10 d	beta (100%)
Protactinium	^{234m}Pa	1.17 m	beta (99.8%), IT [b]
Uranium	^{234}U	$2.45\ 10^{5}$ a	alpha (100%)
Thorium	^{230}Th	$7.54\ 10^{4}$ a	alpha (100%)
Radium	^{226}Ra	1600 a	alpha (100%)
Radon	^{222}Rn	3.824 d	alpha (100%)
Polonium	^{218}Po	3.05 m	alpha (99.98%), beta (0.02%)
Lead	^{214}Pb	26.8 m	beta (100%)
Bismuth	^{214}Bi	19.9 m	alpha (0.02), beta (99.98%)
Polonium	^{214}Po	164 µs	alpha (100%)
Lead	^{210}Pb	22.3 a	beta (100%)
Bismuth	^{210}Bi	5.013 d	beta (100%)
Polonium	^{210}Po	138.4 d	alpha (100%)
Lead	^{206}Pb	stable	
^{232}Th series:			
Thorium	^{232}Th	$1.405\ 10^{10}$ a	alpha (100%)
Radium	^{228}Ra	5.75 a	beta (100%)
Actinium	^{228}Ac	6.15 h	beta (100%)
Thorium	^{228}Th	1.912 a	alpha (100%
Radium	^{224}Ra	3.66 d	alpha (100%)
Radon	^{220}Rn	55.6 s	alpha (100%)
Polonium	^{216}Po	0.145 s	alpha (100%)
Lead	^{212}Pb	10.64 h	beta (100%)
Bismuth	^{212}Bi	60.55 m	alpha (36%), beta (64%)
Polonium	^{212}Po	0.299 µs	alpha (100%)
Thalium	^{208}Tl	3.053 m	beta (100%)
Lead	^{208}Pb	stable	
^{235}U series:			
Uranium	^{235}U	$7.038\ 10^{8}$ a	alpha (100%)
Thorium	^{231}Th	25.52 h	beta (100%)
Protactinium	^{231}Pa	32760 a	alpha (100%)
Actinium	^{227}Ac	21.77 a	alpha (1.4%), beta 98.6%)
Thorium	^{227}Th	18.72 d	alpha (100%)
Francium	^{223}Fr	21.8 m	beta (100%)
Radium	^{223}Ra	11.44 d	alpha (100%)
Radon	^{219}Rn	3.96 s	alpha (100%)
Polonium	^{215}Po	1.781 ms	alpha (100%)
Lead	^{211}Pb	36.1 m	beta (100%)
Bismuth	^{211}Bi	2.14 m	alpha (99.7%), beta 0.3%)
Thalium	^{207}Tl	4.77 m	beta (100%)
Lead	^{207}Pb	stable	

a Electron capture.
b Internal transition.

Table 4
Production rates and concentrations of cosmogenic radionuclides in the atmosphere

Radionuclide	*Production rate*		*Global inventory (PBq)*	*Fractional amount in troposphere* [a]	*Concentration in troposphere* [b] *(mBq m^{-3})*
	Per unit area [a] *(atoms m^{-2} s^{-1})*	*Annual amount* [c] *(PBq a^{-1})*			
^{3}H	2 500	72	1 275	0.004	1.4
^{7}Be	810	1 960	413	0.11	12.5
^{10}Be	450	0.000064	230	0.0023	0.15
^{14}C	25 000	1.54	12 750	0.016	56.3
^{22}Na	0.86	0.12	0.44	0.017	0.0021
^{26}Al	1.4	0.000001	0.71	7.7 10^{-8}	1.5 10^{-8}
^{32}Si	1.6	0.00087	0.82	0.00011	0.000025
^{32}P	8.1	73	4.1	0.24	0.27
^{33}P	6.8	35	3.5	0.16	0.15
^{35}S	14	21	7.1	0.08	0.16
^{36}Cl	11	0.000013	5.6	6 10^{-8}	9.3 10^{-8}
^{37}Ar	8.3	31	4.2	0.37	0.43
^{39}Ar	56	0.074	28.6	0.83	6.5
^{81}Kr	0.01	1.7 10^{-8}	0.005	0.82	0.0012

a References [L5, L6].
b Assumes tropospheric volume of 3.62275 10^{18} m^3; inferred from [L5].
c Assumes surface area of world = 5.1005 10^{14} m^2 [L14].

Table 5
Natural radionuclide content in soil
Data not referenced are from UNSCEAR Survey of Natural Radiation Exposures

Region / country	*Population in 1996 (10^6)*	*Concentration in soil (Bq kg^{-1})*							
		^{40}K		*^{238}U*		*^{226}Ra*		*^{232}Th*	
		Mean	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
Africa									
Algeria	28.78	370	66–1 150	30	2–110	50	5–180	25	2–140
Egypt	63.27	320	29–650	37	6–120	17	5–64	18	2–96
North America									
Costa Rica	3.50	140	6–380	46	11–130	46	11–130	11	1–42
United States [M7]	269.4	370	100–700	35	4–140	40	8–160	35	4–130
South America									
Argentina	35.22	650	540–750						
East Asia									
Bangladesh	120.1	350	130–610			34	21–43		
China [P16, Z5]	1232	440	9–1 800	33	2–690	32	2–440	41	1–360
– Hong Kong SAR [W12]	6.19	530	80–1 100	84	25–130	59	20–110	95	16–200
India	944.6	400	38–760	29	7–81	29	7–81	64	14–160
Japan [M5]	125.4	310	15–990	29	2–59	33	6–98	28	2–88
Kazakstan	16.82	300	100–1 200	37	12–120	35	12–120	60	10–220
Korea, Rep. of	45.31	670	17–1 500						
Malaysia	20.58	310	170–430	66	49–86	67	38–94	82	63–110
Thailand	58.70	230	7–712	114	3–370	48	11–78	51	7–120
West Asia									
Armenia	3.64	360	310–420	46	20–78	51	32–77	30	29–60
Iran (Islamic Rep. of)	69.98	640	250–980			28	8–55	22	5–42
Syrian Arab Republic	14.57	270	87–780	23	10–64	20	13–32	20	10–32

Table 5 (continued)

Region / country	*Population in 1996 (10^6)*	*Concentration in soil (Bq kg^{-1})*							
		^{40}K		*^{238}U*		*^{226}Ra*		*^{232}Th*	
		Mean	*Range*	*Mean*	*Range*	*Mean*	*Range*	*Mean*	*Range*
North Europe									
Denmark [N5]	5.24	460	240–610			17	9–29	19	8–30
Estonia	1.47	510	140–1 120			35	6–310	27	5–59
Lithuania	3.73	600	350–850	16	3–30			25	9–46
Norway	4.35	850		50		50		45	
Sweden	8.82	780	560–1 150			42	12–170	42	14–94
West Europe									
Belgium	10.16	380	70–900			26	5–50	27	5–50
Germany	81.92		40–1 340		11–330		5–200		7–134
Ireland [M6]	3.55	350	40–800	37	8–120	60	10–200	26	3–60
Luxembourg	0.41	620	80–1 800			35	6–52	50	7–70
Netherlands [K2]	15.58		120–730		5–53	23	6–63		8–77
Switzerland	7.22	370	40–1 000	40	10–150	40	10–900	25	4–70
United Kingdom [B2]	58.14		0–3 200		2–330	37			1–180
East Europe									
Bulgaria	8.47	400	40–800	40	8–190	45	12–210	30	7–160
Hungary	10.05	370	79–570	29	12–66	33	14–76	28	12–45
Poland [J7]	38.60	410	110–970	26	5–120	26	5–120	21	4–77
Romania [I12]	22.66	490	250–1 100	32	8–60	32	8–60	38	11–75
Russian Federation	148.1	520	100–1 400	19	0–67	27	1–76	30	2–79
Slovakia	5.35	520	200–1 380	32	15–130	32	12–120	38	12–80
South Europe									
Albania	3.40	360	15–1 150	23	6–96			24	4–160
Croatia	4.50	490	140–710	110	83–180	54	21–77	45	12–65
Cyprus	0.76	140	0–670			17	0–120		
Greece	10.49	360	12–1 570	25	1–240	25	1–240	21	1–190
Portugal	9.81	840	220–1 230	49	26–82	44	8–65	51	22–100
Slovenia	1.92	370	15–1 410			41	2–210	35	2–90
Spain	39.67	470	25–1 650			32	6–250	33	2–210
Median		400	140–850	35	16–110	35	17–60	30	11–64
Population-weighted average		420		33		32		45	

Table 6
External exposure rates calculated from various concentrations of terrestrial radionuclides in soil

Radio-nuclide	*Concentration in soil (Bq kg^{-1})*		*Dose coefficient [I20, S49] (nGy h^{-1} per Bq kg^{-1})*	*Absorbed dose rate in air (nGy h^{-1})*	
	Median value [a]	*Population-weighted value* [a]		*Median value*	*Population-weighted value*
^{40}K	400	420	0.0417	17	18
^{228}U series	35	33	0.462	16	15
^{232}Th series	30	45	0.604	18	27
Total				51	60

a Values from Table 5.

Table 7
External exposure rates from terrestrial gamma radiation
Data not referenced are from UNSCEAR Survey of Natural Radiation Exposures

Region / country	*Population in 1996 (10^6)*	*Absorbed dose rate in air (nGy h^{-1})*				*Ratio indoors to outdoors*
		Outdoors		*Indoors*		
		Average	*Range*	*Average*	*Range*	
Africa						
Algeria [B4]	28.78	70	20–133		14–2 100	
Egypt [H9, I13]	63.27	32	8–93			
Namibia [S12]	1.58					
Sudan	27.29	53	26–690			
North America						
Canada [G3, T14]	29.68	63	43–101			
Cuba [S13]	11.02	42	26–53			
Mexico [C8]	92.72	78	42–140			
United States [M8, O5]	269.4	47	14–118	38	12–160	0.8
South America						
Chile [S14]	14.42	51	21–83	61	25–105	1.2
Paraguay	4.96	46	38–53			
East Asia						
Brunei [L20]	0.30	33	3–70			
China [N3]	1232	62	2–340	99	11–420	1.6
– Taiwan Province [C11]	20	57	17–87			
– Hong Kong SAR [W12]	6.19	87	51–120	200	140–270	2.3
India [N11]	944.6	56	20–1 100			
Indonesia	200.45	55	47–63			
Japan [A7, A8]	125.4	53	21–77	53	21–77	1.0
Kazakstan	16.82	63	10–250	70	20–100	1.1
Korea, Rep. of	45.31	79	18–200			
Malaysia	20.58	92	55–130	96	65–130	1.0
Philippines [D3]	69.28	56	31–120			
Thailand	58.70	77	2–100	48	2–22	0.6
West Asia						
Iran (Islamic Rep. of)	69.98	71	36–130	115	70–165	1.6
Syrian Arab Republic	14.57	59	52–67			
North Europe						
Denmark [N5, S15]	5.24	52	35–70	54	19–260	1.0
Estonia	1.47	59	14–230			
Finland [A9]	5.13	71	45–139	73	22–184	1.0
Iceland [E4]	0.27	28	11–83	23	14–32	0.8
Lithuania	3.73	58	36–85	85	34–195	1.5
Norway [S16, S17]	4.35	73	20–1 200	79	20–1 250	1.1
Sweden [M9]	8.82	56	40–500	110	20–2 000	2.0
West Europe						
Austria [T5]	8.11	43	20–150			
Belgium [D4, S18]	10.16	43	13–80	60	32–90	1.4
France [M10, R3]	58.33	68	10–250	75		1.1
Germany [B5, W11]	81.92	50	4–350	70	13–290	1.4
Ireland [M11, M12]	3.55	42	1–180	62	10–140	1.5
Luxembourg	0.41	49	14–73			
Netherlands [J2, V1]	15.58	32	10–60	64	30–100	2.0
Switzerland	7.22	45	15–120	62	20–200	1.4
United Kingdom [G4, W5]	58.14	34	8–89	60		1.8
East Europe						
Bulgaria [V2]	8.47	70	48–96	75	57–93	1.1
Hungary [N14, N15]	10.05	61	15–130	95	11–236	1.6
Poland [B10, M3]	38.60	45	18–97	67	28–167	1.5
Romania [I12]	22.66	59	21–122	83	30–170	1.4
Russian Federation	148.1	65	12–102	74	24–147	1.1
Slovakia	5.35	67	24–154	79	36–180	1.2
South Europe						
Albania	3.40	71	20–350	100	20–300	1.4
Cyprus	0.76	18	9–52			
Greece	10.49	56	30–109	67	36–131	1.2

Table 7 (continued)

Region / country	*Population in 1996 (10^6)*	*Absorbed dose rate in air (nGy h^{-1})*				*Ratio indoors to outdoors*
		Outdoors		*Indoors*		
		Average	*Range*	*Average*	*Range*	
South Europe						
Italy [B6, C12]	57.23	74	3–228	105	0–700	1.4
Portugal [A10]	9.81	84	4–230	101	4–280	1.2
Slovenia	1.92	56	4–147	75	40–250	1.3
Spain [Q1, Q2]	39.67	76	40–120	110	57–180	1.4
Oceania						
Australia [C13, L7]	18.06	93		103		1.1
New Zealand [R4]	3.6			20	1–73	
Median		57	18–93	75	20–200	1.3 (0.6–2.3)
Population-weighted average		59		84		1.4

Table 8
Outdoor absorbed dose rates in air inferred from concentrations of radionuclides in soil compared with direct measurements

Country	*Absorbed dose rate in air (nGy h^{-1})*		
	From soil concentrations	*From direct measurements*	*Ratio soil/measurements*
Luxembourg	72	49	1.5
Ireland	58	42	1.4
Sweden	77	56	1.4
India	69	56	1.2
China (Hong Kong SAR)	107	87	1.2
Norway	86	73	1.2
United States	55	47	1.2
Switzerland	49	45	1.1
Kazakstan	65	63	1.0
Belgium	44	43	1.0
Portugal	86	84	1.0
Malaysia	93	92	1.0
Egypt	32	32	1.0
Slovenia	56	56	1.0
Romania	58	59	1.0
China	58	62	0.9
Poland	42	45	0.9
Estonia	54	59	0.9
Slovakia	60	67	0.9
Japan	45	53	0.8
Lithuania	48	58	0.8
Thailand	62	77	0.8
Russian Federation	52	65	0.8
Bulgaria	56	70	0.8
Hungary	48	61	0.8
Algeria	54	70	0.8
Iran (Islamic Rep. of)	53	71	0.7
Denmark	39	52	0.7
Spain	54	76	0.7
Greece	39	56	0.7
Albania	40	71	0.6
Syrian Arab Republic	33	59	0.6

Table 9
Distribution of population with respect to the outdoor absorbed dose rate in air from terrestrial gamma radiation

Region / country	*Population (10^3) residing in areas with various levels of outdoor absorbed dose rate in air*											
	< 20 nGy h^{-1}	*20-29 nGy h^{-1}*	*30-39 nGy h^{-1}*	*40-49 nGy h^{-1}*	*50-59 nGy h^{-1}*	*60-69 nGy h^{-1}*	*70-79 nGy h^{-1}*	*80-89 nGy h^{-1}*	*90-99 nGy h^{-1}*	*100-199 nGy h^{-1}*	*200-299 nGy h^{-1}*	*>300 nGy h^{-1}*
East Asia												
Japan		9 619	26 463	20 561	23 382	39 546	5 193					
Korea	1 760	3 096	9 605	4 097	2 220	1 724	4 421	4 421	2 211	11 053		
Malaysia					984	213	1 214	2 498	8 487	6 248		
West Asia												
Iran (Islamic Rep. of)			3 580	1 260	4 896	29 400	4 810	13 080	3 660	3 200		
North Europe												
Denmark		250	2 100	2 200	600	50						
Estonia	6	5	25	149	314	367	592	9				
Finland					913	1 131	2 606	172	325			
Lithuania			138	967	414	1 381	553	276				
West Europe												
Belgium	300	2 200	2 400	2 600	2 500	200	20					
Germany	700	8 600	10 000	20 900	28 000	9 600	1 500	800	700	300		
Italy	125		275	5 300	28 075	8 575	1 975	150	3 100	6 400	3 300	
Luxembourg	29	13	52	230	82	4						
Netherlands	3 459	5 484	2 353	2 976	1 262	47						
Switzerland	60	631	1 131	3 983	584	74	161	74	64	98		
United Kingdom	6 000	12 000	30 000	6 000								
East Europe												
Bulgaria				186	392	5 212	1 239	234	1 614			
Hungary	163	479	836	1 071	1 316	3 488	1 163	765	367	530		17
Poland	150	2 915	12 014	12 268	7 163	2 637	622	213	136			
Romania		295	1 407	4 177	6 447	5 130	3 904	726	568	45		
Russian Federation	450		460	7 150	22 800	84 470	5 730	17 800	5 330	3 910		
Slovakia		22	192	721	1 364	1 292	868	498	243	85		
South Europe												
Albania	0	50	50	100	100	500	2 000	300	200	100	50	50
Greece			1 160	5 605	1 067	1 250	572	147	225	231		
Portugal	333	444	1 814	606	1 325	653	313	582	417	2 352	594	
Spain			1 115	5 490	5 082	10 129	2 903	1 977	2 447	8 152		
Total	13 535	46 103	107 170	108 597	141 282	207 073	42 359	44 722	30 094	42 704	3 944	67
Fraction of total	0.017	0.059	0.14	0.14	0.18	0.26	0.054	0.057	0.038	0.054	0.005	0.00009
Cumulative total	13 535	59 638	166 808	275 405	416 687	623 760	666 119	710 841	740 935	783 639	787 583	787 650
Cumulative fraction	0.017	0.08	0.21	0.35	0.53	0.79	0.85	0.90	0.94	0.995	0.9999	1.000

Table 10
Distribution of population with respect to the average outdoor absorbed dose rate in air from terrestrial gamma radiation

Region / country	*Base population* (10^6)	*Average outdoor dose rate* (nGy h^{-1})	*Fraction of population with respect to decades of outdoor absorbed dose rate in air* [a]												
			-6	*-5*	*-4*	*-3*	*-2*	*-1*	*0*	*1*	*2*	*3*	*4*	*5*	*6*
East Asia															
Japan	124.8	53				0.08	0.21	0.16	0.19	0.32	0.04				
Korea	44.6	79	0.04	0.07	0.22	0.09	0.05	0.04	0.10	0.10	0.05	0.25			
Malaysia	19.6	92			0.05	0.01	0.06	0.13	0.43	0.32					
West Asia															
Iran (Islamic Rep.)	63.9	71			0.065	0.02	0.08	0.46	0.08	0.20	0.057	0.05			
North Europe															
Denmark	5.2	52					0.05	0.40	0.42	0.12	0.01				
Estonia	1.5	59		0.004	0.003	0.02	0.10	0.21	0.25	0.40	0.006				
Finland	5.1	71					0.18	0.22	0.51	0.03	0.06				
Lithuania	3.7	61				0.04	0.26	0.11	0.37	0.15	0.07				
West Europe															
Belgium	10.2	43				0.03	0.22	0.23	0.25	0.24	0.02	0.002			
Germany	81.1	50			0.01	0.12	0.12	0.25	0.34	0.12	0.02	0.01	0.01		
Italy	57.3	74	0.002		0.005	0.09	0.49	0.15	0.03	0.003	0.05	0.11	0.06		
Luxembourg	0.4	43				0.07	0.03	0.13	0.56	0.20	0.01				
Netherlands	15.6	32					0.22	0.35	0.15	0.19	0.08	0.003			
Switzerland	6.9	45				0.009	0.09	0.16	0.58	0.09	0.01	0.02	0.01	0.009	0.01
United Kingdom	54.0	34					0.11	0.22	0.56	0.11					
East Europe															
Bulgaria	8.9	70				0.02	0.04	0.59	0.14	0.03	0.18				
Hungary	10.2	61		0.02	0.05	0.08	0.11	0.13	0.34	0.11	0.08	0.04	0.05		0.002
Poland	38.1	45				0.004	0.08	0.32	0.32	0.19	0.07	0.02	0.006	0.004	
Romania	22.7	59				0.01	0.06	0.18	0.28	0.23	0.17	0.03	0.03	0.002	
Russian Fed.	148.1	65		0.003		0.003	0.05	0.15	0.57	0.04	0.12	0.04	0.03		
Slovakia	5.3	67			0.004	0.04	0.14	0.26	0.24	0.16	0.09	0.05	0.02		
South Europe															
Albania	3.5	71		0.01	0.01	0.03	0.03	0.14	0.57	0.09	0.06	0.03	0.01	0.01	
Greece	10.3	56					0.11	0.55	0.10	0.12	0.06	0.01	0.02	0.02	
Portugal	9.4	84	0.05	0.19	0.06	0.14	0.07	0.03	0.06	0.04	0.25	0.06			
Spain	37.3	76			0.03	0.15	0.14	0.27	0.08	0.05	0.07	0.22			
Total population (10^6)	787.7 [b]		2.3	5.6	17.5	39.8	108	171	232	111	50.8	38.0	9.8	0.53	0.12
Fraction of total			0.003	0.007	0.022	0.051	0.14	0.22	0.29	0.14	0.064	0.048	0.012	0.0007	0.0001

a The average outdoor absorbed dose rate in air in the country is placed in the decade 0; e.g. for 52 nGy h^{-1} average, would indicate decade 0 is the dose interval 50-59 nGy h^{-1}, decade -1 is 40-49 nGy h^{-1}, etc.
b Total distributed population within ranges indicated is 787.3 10^6.

Table 11
Areas of high natural radiation background

Country	*Area*	*Characteristics of area*	*Approximate population*	*Absorbed dose rate in air* [a] *(nGy h^{-1})*	*Ref.*
Brazil	Guarapari	Monazite sands; coastal areas	73 000	90–170 (streets) 90–90 000 (beaches)	[P4, V5]
	Mineas Gerais and Goias			110–1 300	[A17, P4]
	Pocos de Caldas Araxá	Volcanic intrusives	350	340 average 2 800 average	[V5]
China	Yangjiang Quangdong	Monazite particles	80 000	370 average	[W14]
Egypt	Nile delta	Monazite sands		20–400	[E3]
France	Central region Southwest	Granitic, schistous, sandstone area Uranium minerals	7 000 000	20–400 10–10 000	[J3] [D10]
India	Kerala and Madras	Monazite sands, coastal areas 200 km long, 0.5 km wide	100 000	200–4 000 1 800 average	[S19, S20]
	Ganges delta			260–440	[M13]
Iran (Islamic Rep. of)	Ramsar Mahallat	Spring waters	2 000	70–17 000 800–4 000	[S21] [S58]
Italy	Lazio Campania Orvieto town South Toscana	Volcanic soil	5 100 000 5 600 000 21 000 ~100 000	180 average 200 average 560 average 150–200	[C12] [C12] [C20] [B21]
Niue Island	Pacific	Volcanic soil	4 500	1 100 maximum	[M14]
Switzerland	Tessin, Alps, Jura	Gneiss, verucano, ^{226}Ra in karst soils	300 000	100–200	[S51]

a Includes cosmic and terrestrial radiation.

Table 12
Distribution of population with respect to the indoor absorbed dose rate in air from terrestrial gamma radiation

Region / country	*Population (10^3) residing in areas with various levels of indoor absorbed dose rate in air*											
	< 20 nGy h^{-1}	*20-29 nGy h^{-1}*	*30-39 nGy h^{-1}*	*40-49 nGy h^{-1}*	*50-59 nGy h^{-1}*	*60-69 nGy h^{-1}*	*70-79 nGy h^{-1}*	*80-89 nGy h^{-1}*	*90-99 nGy h^{-1}*	*100-199 nGy h^{-1}*	*200-299 nGy h^{-1}*	*>300 nGy h^{-1}*
North Europe												
Denmark		80	300	600	1 100	1 300	1 000	600	200	50	10	
Finland		144	633	779	636	398	474	491	470	1 122		
Lithuania			23	145	239	633	746	798	338	807		
West Europe												
Belgium			600	2 000	2 500	2 500	2 000	620				
East Europe												
Bulgaria					270	3 362	2 370	2 572	201	102		
Hungary	245	581	306	245	357	696	696	903	1 150	4 988	17	16
Romania			23	272	1 498	2 293	5 538	7 038	5 176	794	68	
Russian Federation		460	7 670	20 970	21 020	24 660	12 810	4 860	38 670	16 980		
South Europe												
Greece			329	900	5 536	1 043	1 135	417	293	603		
Italy	800	2 175	4 225	5 400	4 400	3 975	4 850	8 500	6 800	8 875	4 025	3 250
Spain					1 115	5 490	1 285	8 908	4 015	15 584	906	
Total	1 045	3 440	14 109	31 311	38 671	46 350	32 904	35 707	57 313	49 905	5 026	3 266
Fraction of total	0.003	0.011	0.044	0.10	0.12	0.15	0.10	0.11	0.18	0.16	0.016	0.010
Cumulative total	1 045	4 485	18 594	49 905	88 576	134 927	167 831	203 537	260 851	310 755	315 781	319 047
Cumulative fraction	0.003	0.014	0.058	0.16	0.28	0.42	0.53	0.64	0.82	0.97	0.99	1.000

Table 13
Reference annual intake of air, food, and water
[I7, W1]

Intake	*Breathing rate (m^3 a^{-1})*		
	Infants (1 year)	*Children (10 years)*	*Adults*
Air	1 900	5 600	7 300
Intake	*Food consumption rate (kg a^{-1})*		
	Infants	*Children*	*Adults*
Milk products	120	110	105
Meat products	15	35	50
Grain products	45	90	140
Leafy vegetables	20	40	60
Roots and fruits	60	110	170
Fish products	5	10	15
Water and beverages	150	350	500

Table 14
Concentrations of uranium and thorium series radionuclides in air

Region / country	*Concentration (µBq m^{-3})*									*Ref.*
	^{238}U	^{230}Th	^{226}Ra	^{210}Pb	^{210}Po	^{232}Th	^{228}Ra	^{228}Th	^{235}U	
North America United States	0.9–5	0.6	0.6	100-1 000	10-40	0.4		1.0	0.04	[F3, L8, M15, P5, W6]
Europe Germany Netherlands Norway Poland Switzerland	0.3–1.7 0.02–0.06 1–18	0.3–1.7 0.02–0.07	1.2–3.3 0.8–32	28–2 250 410 <40–710 200-2 000	12-80	0.2–0.9 0.01–0.07	<0.3-1.5			[H31, K4, K10] [N21] [K4] [K5] [S51]
Reference value	1	0.5	1 (0.5) [a]	500	50	0.5 (1)[a]	1	1	0.05	[U3]

a Revised value; previous value [U3] in parentheses.

Table 15
Concentrations of uranium and thorium series radionuclides in foods and drinking water

Region / country	*Concentration (mBq kg^{-1})*									*Ref.*
	^{238}U	^{230}Th	^{226}Ra	^{210}Pb	^{210}Po	^{232}Th	^{228}Ra	^{228}Th	^{235}U	
Milk products										
North America										[F3, M16]
United States	0.7	0.4	5.7	11		0.27			0.05	
Asia										
China	13		6	16	13	1.2	21		0.6	[Z1]
India	17				15					[D6, K6]
Japan	0.55		12			0.29				[S22]
Europe										
Italy			3–19							[M17]
Germany			2–130	5–280	2–80					[B3, J4, M18]
Poland	2.6	1.2	10	18	16	1.2				[P3, P7]
Romania			0.9–44	11–15	13–140					[B20,R20]
U.K.	0.1–4.9		<0.4–200	35–88	20–220			56		[B2]
Reference value	1	0.5	5	15 (40) [a]	15 (60)	0.3	5	0.3	0.05	
Meat products										
North America										
United States	0.8–2.3	0.5–3	20	18		0.3–2			0.02	[F3,M16]
Asia										
China	10		41	140	120	4.3	120		0.5	[Z1]
India					440					[K6]
Japan	13		36			2.3				[S22]
Europe										
Germany	1–20		30–220	100–1 000	37-4 000					[B3, G5,M18]
Poland	1.6–5.6	0.7–3.0	11–19	98–105	99–102	0.5–3.6				[P3, P7]
Romania			2–30	15–19	38–110					[B20,R20]
U.K.	4.9		2.6–74	40-3 700	62-67 000			22–93		[B2]
Reference value	2	2	15	80	60	1	10	1	0.05	
Grain products										
North America										
United States	3–23	0.9–10	7–100	33–81		0.1–2.8			0.1–1.3	[F3,M16]
Asia										
China	9.8		17	34	42	13	38		0.5	[Z1]
India	7.4–67				15–120					[D6, K6]
Japan	1.2		14			1.2				[S22]
Europe										
Germany	20–400		20-2 900	40-4 000	37–1 900					[B3, G5]
Poland	4.7–11	1.4–17	80–110	110–160	90–140	2.0–21				[P3, P7]
Romania	6.1–85		30–90	49–59	20–360	1.6–33				[B20,R20]
U.K.	6.2–35		0.7-5 200	56–120	27–260	12		180-2300		[B2]
Reference value	20	10	80	50 (100)	60 (100)	3	60	3	1	
Leafy vegetables										
North America										
United States	24	20	56	41		18			1.2	[F3,M16]
Asia										
China	16		75	360	430	23	220		0.7	[Z1]
India	61–72				320					[D6, K6]

Table 15 (continued)

Region / country	*Concentration (mBq kg^{-1})*									*Ref.*
	^{238}U	^{230}Th	^{226}Ra	^{210}Pb	^{210}Po	^{232}Th	^{228}Ra	^{228}Th	^{235}U	
Europe Germany Italy Poland U.K.	 6–2 200 14–15 9.8–400	 6–9 80–380	 6–1 150 27–44 37–43 2.2–170	 4–4 100 43–51 16-3 300	 4–7 400 40–67 37-3 300	 4–7				[B3, G5,M18] [D9] [P3, P7] [B2]
Reference value	20	20	50	80 (30)	100 (30)	15	40	15	1	
Root vegetables and fruits										
North America United States	 0.9–7.7	 0.2–1.1	 7–47	 8–150		 0.08–1.4			 0.1	 [F3,M16]
Asia China India Japan	 13 0.4–77 26		 63 11	 27	 29 16–140	 4.7 2.3	 110		 0.6	 [Z1] [D6, K6] [S22]
Europe Germany Italy Poland Romania U.K.	 10-2 900 0.9–10 6–120 6	 0.7–7.5	 5–9 400 14–25 11–215 9–190 9.0–41	 20-4 900 24–93 19–44 18–76	 22–5 200 28–210 12–140	 0.7–7.1 0.4–2.1		 22		 [B3, G5,M18] [D9] [P3, P7] [B20,R20] [B2]
Reference value	3	0.5	30	30 (25)	40 (30)	0.5	20	0.5	0.1	
Fish products										
North America United States	 13-1 900	 1.2–29	 30–59	 14-1 800	 150-55 000	 1.2–30			 0.4-90	 [F3, M16, S23]
Asia China	 12		 39	 3 500	 4 900	 1.3	 320		 0.5	 [Z1]
Europe France Germany Poland Portugal U.K.	 2.5		 37 100-7 400 28–43 8.5-2 100	 20-4 400 81–93 180-4 800	 50-5 200 3 100-3 800 80-120 000 60-53 000			 56–700		 [P6] [G5,M18] [P7] [C14] [B2]
Reference value	30	10	100	200	2 000	10		100		
Drinking water										
North America United States	 0.3–77	 0.1	 0.4–1.8	 0.1–1.5		 0.05	 0–0.5		 0.04	[C15, F3, H11, M20]
Asia China India	 0.1–700 0.09–1.5		 0.2–120			 0.04–12				 [N3] [D6]
Europe Finland France Germany Italy Poland Romania Switzerland Spain U.K.	 0.5– 150 000 4.4–930 0.4–600 0.5–130 7.3 0.4–37 0–1 000 3.7–4.4	 1.4	 10– 49 000 7–700 1–1 800 0.2–1200 1.7–4.5 0.7–21 0–1500 <20-4 000 0–180	 0.2– 21 000 0.2–200 1.6 7–44 40–200	 0.2–7 600 0.1–200 0.5 7–44	 0–4.2 0.06 0.04–9.3	 18-570 0–200	 	 0–50	 [A16,S11] [D8, P6] [B3, G5, G6] [S55] [P3, P7] [B20,R20] [S51] [S24] [B2]
Reference value	1	0.1	0.5	10	5	0.05	0.5	0.05	0.04	

a Revised value with previous value [U3] (if different) in parentheses.

Table 16
Annual intake of uranium and thorium series radionuclides in diet

Region / country	*Annual intake (Bq)*									*Ref.*
	^{238}U	^{230}Th	^{226}Ra	^{210}Pb	^{210}Po	^{232}Th	^{228}Ra	^{228}Th	^{235}U	
North America										
Puerto Rico			9.1							[H13]
United States	5.5–6.2	2.2–3.7	10–24	16–23	22	1.1–2.2	13–16	7.3–8.0		[B8, F3, F5, H12, L9, M23, M24, P8, S31, S33, W6]
South America										
Argentina			9.5		18					[B7, U8]
Brazil			40				40			[L10, P9]
Asia										
China	57		12–32	75–110	68–130	9.3	66		2.6	[L16,Y5,Y6, Z1]
India	2.9		8.8	46	20	3.3	47			[C16, D5, L11, S41]
Japan	3.2–6.6	0.6	9–15	73–80	220	0.6–0.8				[K7, N13, S22, S27, S42, S45]
Europe										
Belgium			16							[S28]
Bulgaria				22–28						[K8]
Czech Rep.			40							[T6]
France	4.4		14–19	18						[G7, S32, U8]
Germany	11	3.4	40	62		2.2	17	17		[F4, G8, M22]
Italy			11–19	40	40					[C17, D9, M21]
Netherlands			27							[S28]
Poland	6.4	2.1	19–20	45	44	1.2				[P3, P7]
Romania	5.8		19	57	51	2.2		2.2		[I12]
Russian Fed.	16			55–84	40–55					[D7, L12]
U.K.	4.4		11–16	16–30	28–44					[C18, H14, H15, S29, S30]
Reference value [a]	5.7	3.0	22	30	58	1.7	15	3.0	0.2	

a Intake by adults; consumption rates from Table 13 and reference concentrations in foods and water from Table 15.

Table 17
Annual effective dose from inhalation of uranium and thorium series radionuclides

Radio-nuclide	*Concentration in air ($\mu Bq\ m^{-3}$)*	*Effective dose coefficient [I9] ($\mu Sv\ Bq^{-1}$)*			*Committed effective dose [a, b] (μSv)*			
		Infants	*Children*	*Adults*	*Infants*	*Children*	*Adults*	*Age-weighted*
^{238}U	1	9.4	4	2.9	0.018	0.022	0.021	0.021
^{234}U	1	11	4.8	3.5	0.021	0.027	0.026	0.026
^{230}Th	0.5	35	16	14	0.033	0.045	0.051	0.048
^{226}Ra	1	11	4.9	3.5	0.021	0.027	0.026	0.026
^{210}Pb	500	3.7	1.5	1.1	3.5	4.2	4.0	4.0
^{210}Po	50	11	4.6	3.3	1.0	1.3	1.2	1.2
^{232}Th	0.5	50	26	25	0.048	0.073	0.091	0.084
^{228}Ra	1	10	4.6	2.6	0.019	0.026	0.019	0.021
^{228}Th	1	130	55	40	0.25	0.31	0.29	0.29
^{235}U	0.05	10	4.3	3.1	0.001	0.001	0.001	0.001
Total					5.0	6.0	5.8	5.8

a Assumed breathing rates: infants 1,900 $m^3\ a^{-1}$, children 5,600 $m^3\ a^{-1}$, adults 7,300 $m^3\ a^{-1}$.
b Committed effective dose from the annual intake. Age distribution for weighted values: infants 0.05, children 0.3, adults 0.65.

Table 18
Annual intake and effective dose from ingestion of uranium and thorium series radionuclides

Radio-nuclide	*Activity intake [a] (Bq)*			*Effective dose coefficient [I2, I21] ($\mu Sv\ Bq^{-1}$)*			*Committed effective dose [b] (μSv)*			
	Infants	*Children*	*Adults*	*Infants*	*Children*	*Adults*	*Infants*	*Children*	*Adults*	*Age-weighted*
^{238}U	1.9	3.8	5.7	0.12	0.068	0.045	0.23	0.26	0.25	0.25
^{234}U	1.9	3.8	5.7	0.13	0.074	0.049	0.25	0.28	0.28	0.28
^{230}Th	1.0	2.0	3.0	0.41	0.24	0.21	0.42	0.48	0.64	0.58
^{226}Ra	7.8	15	22	0.96	0.80	0.28	7.5	12	6.3	8.0
^{210}Pb	11	21	30	3.6	1.9	0.69	40	40	21	28
^{210}Po	21	39	58	8.8	2.6	1.2	180	100	70	85
^{232}Th	0.6	1.1	1.7	0.45	0.29	0.23	0.26	0.32	0.38	0.36
^{228}Ra	5.5	10	15	5.7	3.9	0.69	31	40	11	21
^{228}Th	1.0	2.0	3.0	0.37	0.15	0.072	0.38	0.30	0.22	0.25
^{235}U	0.1	0.2	0.2	0.13	0.071	0.047	0.011	0.012	0.012	0.011
Total							260	200	110	140

a Consumption rates from Table 13 and concentrations in foods and water (reference values) from Table 15.
b Committed effective dose from the annual intake. Age distribution for weighted values: infants 0.05, children 0.3, adults 0.65.

Table 19
Uranium and thorium series radionuclides in human tissues

Region / country	*Concentration (mBq kg^{-1})*					*Ref.*
	Lung	*Liver*	*Kidney*	*Muscle and other tissues*	*Bone* [a]	
^{238}U						
Africa Nigeria					340	[F9]
North America Canada United States	6.2–15	1.5–4.1	4.8–12		120 11–52	[F9] [F8, S44]
South America Brazil					130–150	[F9]
East Asia China India Japan Nepal	21	3.0	27 4.2	5.3	410 (94–2 600) 140 17–59 110	[L1] [G13] [I17] [F10]
Europe Austria United Kingdom Yugoslavia		3.1	62	2.4	10 150 2.7	[H20] [H15] [P14]
Russian Federation	67–84	72–140	66–68	81–95	74–120	[D7, F10, M31]
Oceania Australia					23	[F10]
Median value Range	21 (6–84)	3 (2–140)	27 (4–68)	5 (2–95)	100 (3–410)	
Reference value	20 (15) [b]	3	30 (5)	5 (2)	100 (50)	
^{230}Th						
Africa Nigeria					110	[F11]
North America Canada United States	12–31	6	6–11		41 45–130	[F11] [H23, I15, S1]
East Asia China Japan	29 19	12	1	1.4	120 (58–220) 24	[C3] [H22]
Median value Range	19 (12–29)	9 (6–12)	5 (1–11)	1	76 (24–120)	
Reference value	20	9 (7)	5 (10)	1 (0.3)	20–70 [c]	
^{226}Ra						
16 countries [d] 31 countries [e]	3.6 4.1	3.6 4.1	3.6 4.1	3.6 4.1	230 260	[F15, U7] [F16]
Reference value	4.1 (2.7)	4.1 (2.7)	4.1 (2.7)	4.1 (2.7)	260 (170)	
^{210}Pb						
Europe Finland Russian Federation	240	90 450	170 270	30 140–270	2 400 5 000	[K17] [L12]
East Asia Japan	240	560	430	30–230	2 600	[T13]
North America United States	230	340	160	140		[B22]

Table 19 (continued)

Region / country	*Concentration (mBq kg^{-1})*					*Ref.*
	Lung	*Liver*	*Kidney*	*Muscle and other tissues*	*Bone* [a]	
Median value Range	240 (230–240)	400 (90–560)	220 (160–430)	110 (30–270)	2 600 (2 400–5 000)	
Reference value	200	400 (200)	200	100 (200)	3 000	
^{210}Po						
Europe Finland Russian Federation United Kingdom	 330 200	 510 970 630	 490 760 640	 110 110–220 120	 2 200 2 400 2 200	 [K17] [L12] [H10]
East Asia Japan	370	1 700	1 200	40–310	2 600	[T13]
North America United States	190	410–540	420	130–220	2 900	[B22, H30]
Median value Range	270 (190–370)	630 (410–970)	640 (420–1 200)	120 (40–310)	2 400 (2 200–2 900)	
Reference value	200 (100)	600 (200)	600 (200)	100 (200)	2 400	
^{232}Th						
Africa Nigeria					86	[F11]
North America Canada United States	 9.3–32	 2.2–3.0	 1.9–4.1		15 21–35	[F11] [H23, I15, L15, S1, W7]
East Asia China India Japan	 38 24 22	 3.6 2.1	 6.8 1.0	 2.2 0.8	 68 (34–140) 8 11	 [C3] [J8, S41] [H22]
Europe United Kingdom Yugoslavia	 22				 62 50	 [H21] [P14]
Median value Range	22 (9–53)	3 (2–4)	3 (1–7)	1 (1–2)	38 (8–86)	
Reference value	20	3 (2)	3	1 (0.15)	6–24 [c]	
^{228}Ra						
Africa Nigeria					320	[F11]
North America Canada United States	 9–10	 2.6–3.3	 2.6–3.3		 23 39–230	 [F11] [I15, S1]
East Asia China Japan	 41 19	 3.9	 1.3	 1.5	 290 (140–570) 100	 [C3] [H22]
Median value Range	19 (9–41)	3 (3–4)	2 (1–3)	2	100 (23–320)	
Reference value	20 (15)	3 (5)	2 (10)	2 (0.5)	100 (50)	

a Assumes 5 kg dry bone yields 2.7 kg ash per skeleton.
b Revised reference value with previous value [U4] (if different) in parentheses.
c First value given is for cortical bone and the second value for trabecular bone.
d Representing 30% of the world population.
e Representing 66% of the world population.

Table 20
Dose rates to adults from ingestion of uranium and thorium radionuclides estimated from reference concentrations in tissues

Tissue	*Concentration (mBq kg^{-1})*					
	$^{238/234}U$	^{230}Th	^{226}Ra	$^{210}Pb/Po$	^{232}Th	$^{228/224}Ra$
Bone	100	30 [b]	260	2 400 [c]	9.6[b]	100
Lung	20	20	4.1	200	20	20
Kidney	30	5	4.1	600	3	2
Liver	3	9	4.1	600	3	3
Other [a]	5	1	4.1	100	1	2
	Absorbed dose rate per unit concentration (μGy a^{-1} per mBq kg^{-1}) [U4]					
	$^{238/234}U$	^{230}Th	^{226}Ra [d]	$^{210}Pb/Po$	^{232}Th	$^{228/224}Ra$
Soft tissues	0.046	0.024	0.063	0.027	0.020	0.16
Bone marrow	0.085	1.9	0.18	0.046	1.1	0.70
Bone lining cells [e]	0.008	0.005	0.022	0.005	0.003	0.056
	Effective dose rate (μSv a^{-1})					
	$^{238/234}U$	^{230}Th	^{226}Ra	$^{210}Pb/Po$	^{232}Th	$^{228/224}Ra$
Body [f]	7	6	7	80	4	18
Total [f]	120					
Intake [g]	0.5	0.6	6	91	0.4	11
Total [g]	110					

a Includes gonads, breast, red bone marrow, and thyroid.
b Assumes 4 kg cortical and 1 kg trabecular bone in 5 kg skeleton.
c Concentration of ^{210}Po.
d Includes dose from ^{222}Rn and its short-lived decay products; retention factor of one third.
e Referred to concentration in bone; cells located 10 μm from bone surface [H23].
f Estimated from reference concentrations in body.
g Estimated from intake of radionuclides in foods and water for adults (see Table 18).

Table 21
Parameters of the model masonry house

Parameter	*Notation*	*Value*
Dimensions and relevant parameters		
Volume		250 m^3
Surface area of floor		100 m^2
Length of floor to wall gap		40 m
Width of floor to wall gap		3 mm
Total surface area including internal walls, furniture, etc.		450 m^2
Air exchange rate		1 h^{-1}
Subsoil		
Activity concentration of ^{226}Ra	C_{Ra}	50 Bq kg^{-1}
Emanation fraction	f	0.2
Porosity	ε	0.25
Fraction of water saturation	m	0.2
Effective diffusion coefficient	D_e	2.0 10^{-6} m^2 s^{-1}
Bulk diffusion coefficient	D	5.0 10^{-7} m^2 s^{-1}
Soil density	ρ	1 600 kg m^{-3}
Permeability	k	2 10^{-11} m^2
Aggregate layer thickness [a]		0.15 m
Aggregate layer permeability [a]		5 10^{-9} m^2
Building elements, wall and floor		
Thickness of floor		0.1 m
Thickness of walls and ceiling		0.2 m
Activity concentration of ^{226}Ra	C_{Ra}	50 Bq kg^{-1}
Emanation fraction	f	0.1
Porosity of wall	ε	0.15
Porosity of floor		0.20
Effective diffusion coefficient of wall	D_e	7 10^{-8} m^2 s^{-1}
Effective diffusion coefficient of floor		1 10^{-7} m^2 s^{-1}
Density	ρ	1 600 kg m^{-3}

a The other specifications of the aggregate layer are similar to the subsoil specifications.

Table 22
Representative radon entry rates of the model masonry house

Source of radon		*Radon entry rate (Bq m^{-3} h^{-1})*
Building materials	Diffusion, walls and ceiling	10
	Diffusion, floor slab	1
Subjacent earth	Diffusion through the slab	10
	Diffusion through gaps	4
	Advection	20
Outdoor air	Infiltration	10
Water supply	De-emanation	1
Natural gas	Consumption	0.3
Total		56

Table 23
Representative radon entry rates in low-rise residential houses in Finland compared to the model masonry house

Source of radon	*Mechanism*	*Radon entry rate (Bq m^{-3} h^{-1})* [a]		
		Wooden house [b]	*Masonry house* [c]	*Model masonry house* [d]
Building elements				
Walls and ceiling	Diffusion	2 (3)	16 (18)	10 (18)
Floor slab	Diffusion			1 (2)
Subjacent earth				
Through gaps	Diffusion			4(7)
	Advection	60 (86)	66 (73)	20 (35)
Through slab	Diffusion	4 (6)	4 (4)	10 (18)
Outdoor air	Infiltration	3(4)	3 (3)	10 (18)
Water supply	De-emanation	1 (1)	1 (1)	1 (2)
Natural gas	Consumption			0.3 (–)
Total		70 (100)	90 (100)	56 (100)

a Percentage in parentheses.
b Radon concentration indoors 140 Bq m^{-3}; air exchange rate 0.5 h^{-1} [A1].
c Radon concentration indoors 180 Bq m^{-3}; air exchange rate 0.5 h^{-1} [A1].
d Radon concentration indoors 56 Bq m^{-3}; air exchange rate 1.0 h^{-1} [U3].

Table 24
Radon concentrations in dwellings determined in indoor surveys
Data not referenced are from UNSCEAR Survey of Natural Radiation Exposures

Region	*Country*	*Population in 1996 (10^6)*	*Radon concentration (Bq m^{-3})*			*Geometric standard deviation*	*Ref.*
			Arithmetic mean	*Geometric mean*	*Maximum value*		
Africa	Algeria	28.78	30		140		[C19]
	Egypt	63.27	9		24		[K12]
	Ghana	17.83			340		[O6]
North America	Canada	29.68	34	14	1 720	3.6	[L13]
	United States	269.4	46	25		3.1	[M26, U14]
South America	Argentina	35.22	37	26	211	2.2	[G9]
	Chile	14.42	25		86		[S14]
	Paraguay	4.96	28		51		
East Asia	China	1232	24	20	380	2.2	[Z2]
	– Hong Kong SAR	6.19	41		140		[T8]
	India	944.6	57	42	210	2.2	[S37]
	Indonesia	200.45	12		120		
	Japan	125.4	16	13	310	1.8	[F20, S61]
	Kazakstan	16.82	10		6 000		
	Malaysia	20.58	14		20		
	Pakistan	140.0	30		83		[T9]
	Thailand	58.7	23	16	480	1.2	
West Asia	Armenia	3.64	104		216	1.3	
	Iran (Islamic Rep. of)	69.98	82		3 070		[S38]
	Kuwait	1.69	14	6	120		[B15]
	Syria	14.57	44		520		[O8]
North Europe	Denmark	5.24	53	29	600	2.2	[S15, U15]
	Estonia	1.47	120	92	1 390		[P15]
	Finland	5.13	120	84	20 000	2.1	[A13, C21]
	Lithuania	3.73	55	22	1 860		
	Norway	4.35	73	40	50 000		
	Sweden	8.82	108	56	85 000		[S25]
West Europe	Austria	8.11		15	190		[S34, S35]
	Belgium	10.16	48	38	12 000	2.0	
	France	58.33	62	41	4 690	2.7	[R5]
	Germany	81.92	50	40	>10 000	1.9	
	Ireland	3.55		37	1 700		[C22]
	Luxembourg	0.41	110	70	2 500	2.0	
	Netherlands	15.58	23	18	380	1.6	[N22, P10]
	Switzerland	7.22	70	50	10 000		[S26]
	United Kingdom	58.14	20		10 000		[W5]
Eastern Europe	Bulgaria	8.47		22	250		
	Czech Republic	10.25	140		20 000		[T7]
	Hungary	10.05	107	82	1 990	2.7	[N14]
	Poland	38.60	41	32	432	2.0	[B10]
	Romania	22.66	45		1 025		[I12]
	Slovakia	5.35	87		3 750		
South Europe	Albania	3.40	120	105	270	2.0	
	Croatia	4.50	35	32	92		
	Cyprus	0.76	7	7	78	2.6	[C6]
	Greece	10.49	73	52	490		[G10]
	Italy	57.23	75	57	1 040	2.0	[B9]
	Portugal	9.81	62	45	2 700	2.2	[F7]
	Slovenia	1.92	87	60	1 330	2.2	[K15]
	Spain	39.67	86	42	15 400	3.7	
Oceania	Australia	18.06	11	8	420	2.1	[L7]
	New Zealand	3.60	20	18	90		[R4]
Median			46	37	480	2.2	
Population-weighted average			39	30	1 200	2.3	

Table 25
Thoron concentrations in outdoor and indoor air

Region / country or territory	*Equilibrium equivalent concentration [a] ($Bq\ m^{-3}$)*		*$^{220}Rn/^{222}Rn$ EEC ratio*		*Ref.*
	Outdoors	*Indoors*	*Outdoors*	*Indoors*	
North America					
United States		0.5 (0.03–4.7)		0.04	[T11]
	0.09 (0.03–0.3)	0.2 (0.1–0.3)			[H36]
East Asia					
China	0.4	0.8	0.05	0.07	[P12]
Hong Kong SAR	0.3 (0.1–0.5)	0.8 (0.4–1.2)	0.04	0.06	[T10]
Japan		0.6 (0.4–0.9) [b]		0.1	[G12, G23]
		0.5 (0.1–1.0) [b]			[Y8]
		0.2 (0.1–0.6) [b]		0.03	[Z6]
		3.2 (1.0–6.0) [c]		0.4	[G12, G23]
		2.7 (0.2–8.2) [c]			[Y8]
		1.7 (0.3–5.3) [c]		0.2	[Z6]
	0.09 (0.03–0.12)	0.7 (0.04–2.1) [d]			[Y8]
		1.5 (1.4–1.6) [d]			[G23]
Malaysia	0.5 (0.3–1.8)	1.1 (0.4–2.5)	0.08	0.08	[e]
North Europe					
Norway		0.7 (0.07–1.1)		0.04	[S43]
Sweden		0.3 (0.1–0.6)		0.01	[M29]
West Europe					
France		0.8 (0.6–13.3)		0.03	[R10]
United Kingdom		0.3 (0.07–1.1)		0.02	[C26]
Central Europe					
Germany		0.5 (0.1–1.0)			[P13]
Rep. of Moldova	0.2	1.0 (0.1–6.4)	0.04	0.05	[I16]
Romania	0.3 (0.1–0.6)	1.1 (0.1–6.4)	0.05	0.04	[I12, M30]
East Europe					
Russian Federation		1.1–7.1		0.09 (0.02–0.24)	[Z4]
South Europe					
Italy		12 (0.5–76)		0.11 (0.01–0.38)	[B14, S7, S9]
Slovenia	0.12 (0.05–0.37)		0.013		[K3]
Range	0.09–0.5	0.2–12	0.01–0.08	0.01–0.5	

a Range in parentheses.
b Concrete dwellings.
c Wood frame and mud dwellings.
d New materials, e.g. mortar wallboard.
e Response to UNSCEAR Survey of Natural Radiation Exposures.

Table 26
Principal dosimetric assessments of lung dose from deposited radon decay products

Year	*Investigator*	*Parameter values*		*Target region*	*Model type*	*Dose factor* [a] *[nGy (Bq h m^{-3})$^{-1}$]*
		Unattached fraction	*Breathing rate (m^3 h^{-1})*			
1956	Chamberlain, Dyson [C9]	0.09	1.2	Average in 45 µm epithelium	Cast of trachea and bronchi	11
1959	ICRP [I19]	0.1	1.2	Mean TB region	Deposition retention assumptions	6.7
1964	Jacobi [J10]	0.25		Basal cells (30 µm)	Findeisen/Landahl 6-region anatomical model	24
1964	Altshuler et al. [A6]	0.085	0.9	Basal cells (22 µm)	Findeisen/Landahl 6-region anatomical model	32
1967	Haque, Collinson [H24]	0.35		Basal cells (30 µm)	Weibel dichotomous airway model	71
1972	Harley, Pasternack [H25]	0.04	0.9	Basal cells (22µm)	Weibel dichotomous airway model	5.7
1980	Jacobi, Eisfeld [J5]	0.1	1.2	Mean epithelium	Weibel dichotomous airway model, correction for upper airway turbulent diffusion [M19]	8.9
1980	James et al. [J6]	0.1	1.2	Mean epithelium	Yeh-Shum anatomical model [Y4]	14
1982	Harley, Pasternack [H26]	0.07	1.1	Basal cells (22 µm)	Weibel dichotomous airway model, correction for upper airway turbulent diffusion [M19]	6.4
1982	Hoffman [H27]	0.2	0.9	Mean epithelium	Weibel dichotomous airway model, correction for upper airway turbulent diffusion [M19]]	11
1991	National Research Council [N17]	0.16	1.2	Basal cells (35–50 µm)	Yeh-Shum anatomical model [Y4], correction for upper airway turbulent diffusion	21
1996	Harley et al. [H8]	0.1	1.2	Basal cells (27 µm)	Nikiforov et al. [N9] anatomical model, airway deposition from empirical data from human airway casts	9
1998	Marsh, Birchall [M25]	0.08	0.8	Bronchial cells Basal (35–50 µm) Secretary (10–40 µm) Bronchiolar cells Secretary (4–12 µm)	ICRP lung model [I7]	 8.5 19 14

a Per unit ^{222}Rn concentration (EEC). WLM converted to Bq h m^{-3} with 0.27 10^{-3} WL (Bq m^{-3})$^{-1}$ and 170 h per working month.

Table 27
Typical concentrations of radionuclides in raw and produced materials and in wastes of the mineral processing industry

Material	*Typical concentration in ore / raw material (kBq kg^{-1})*		*Typical concentration in product or tailings / wastes (kBq kg^{-1})*	
	^{238}U-series	*^{232}Th-series*	*^{238}U-series*	*^{232}Th-series*
Phosphate industry				
Phosphate Artificial fertilizer	0.2–1.5 1.5 (Florida ore) 0.03 (Kovdor ore) 0.11 (Palfos ore) 0.3–3 0.2–1 (^{226}Ra and ^{210}Pb) 2.2 (TSP)	0.02 (Florida ore) 0.008–0.04 0.005 (TSP)	0.9–1.3 [a] 100 (^{210}Po) 600 (^{210}Pb) in calcinate 1 (Phosphorus slag)	0.02 (Phosphorus slag)
Rare earths, thorium compounds				
Monazite	6–40	4% (by weight) 8–300	450 [b]	3000 [b]
Oil and gas extraction				
Natural gas Oil	0.34 kBq m^{-3} (^{222}Rn)		(Scale) 1–1 000 (scale) 8–42 kBq m^{-3} (production water)	
Metal ores				
Iron ore Cassiterite Pyrochlore	 1 6–10	 0.3 7–80	0.1–0.3 (coal tar) 0.15 (blast furnace slag) / zinc-rich filtercake 1 (slag)	0.15 (blast furnace slag) 4 (slag)
Coal tar treatment				
Coal tar	0.1–0.3 (^{210}Po and ^{210}Pb)		0.2–0.6 (electrode pitch)	
Cokes and electric power production				
Coal	0.01–0.025	0.01–0.025	0.02–0.04 (cokes) 0.1–0.3 (coal tar) 0.2 (fly and bottom ash) 0.4 (fly dust)	0.2 (fly dust)
Cement industry				
Marl Schist Portland clinker	0.022 0.04 0.08	0.003 0.056 0.05	0.05–0.11 (cement) 0.02 (silex)	0.03–0.1 (cement) 0.003 (silex)
Mineral sands handling				
Zirconium sand Bauxite Ilmenite Rutile	0.2–74 0.4–0.6 2.3 (1.5 : ^{238}U) 3.8	0.4–40 0.3–0.4 1.2 0.56	Rutile	
Titanium pigment production				
Ilmenite Titanium ore	2.3 (1.5 : ^{238}U) 0.07–9	1.2 0.07–9	400 [c] 0.15 (VBM) 2.3 (filtercake) 0.03 (water)	up to 1 500 (scale) 0.13 (VBM) 2.6 (filtercake) 0.01 (water)

a Phosphogypsum, Central Florida ore.
b ^{226}Ra in sulphate precipitate.
c ^{226}Ra precipitate.

Table 28
Release of radionuclides from typical installations of mineral processing industries
[L18]

Industry	*Ore throughput (kt a^{-1})*	*Releases to atmosphere (GBq a^{-1})*							*Releases to water (GBq a^{-1})*						
		^{238}U	^{228}Th	^{226}Ra	^{222}Rn	^{210}Pb	^{210}Po	^{40}K	^{238}U	^{228}Th	^{226}Ra	^{222}Rn	^{210}Pb	^{210}Po	^{40}K
Elementary phosphorus	570				563	66	490		-	-	-	-	24	166	-
– Transport		0.06	0.001	0.06	0.03	0.06	0.06	0.004	0.18	0.002	0.18	0.18	0.18	0.18	0.013
Phosphoric acid	700	0.07	0.002	0.09	820	0.08	0.14	0.008	336	8	737	-	654	997	79
Fertilizer plant	375				221	0.044	0.034		-	-	-	-	0.054	0.057	-
– Transport		0.02	0.0001	0.02	0.02	0.02	0.02	0.001	0.07	0.0004	0.07	0.07	0.07	0.06	0.002
Iron and steel production	7 500				180	55	90		-	-	-	-	0.51	8	-
– Transport		0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.03	0.03	0.03	0.03	0.04	0.04	0.04
Coal tar treatment	120					~0	~0		-	-	-	-	-	-	-
Coal-fired power plant (600 MW e)	1 350	0.16	0.08	0.11	34	0.4	0.8	0.27	-	-	-	-	-	-	-
– Transport		0.004	0.004	0.004	0.004	0.004	0.004	0.012	0.011	0.011	0.011	0.011	0.011	0.011	0.036
Cokes production	885	0.013	0.009	0.013	13	0.012	0.07	0.032	-	-	-	-	0.024	0.032	-
– Transport		0.001	0.001	0.001	0.001	0.002	0.001	0.004	0.004	0.003	0.004	0.004	0.005	0.004	0.011
Cement industry	2 000	0.2	0.05	0.2	157	0.2	78	0.4	-	-	-	-	-	-	-
Ceramics	3 200	0.03		0.03	0.03	0.09	0.3	0.14	-	-	-	-	-	-	-
Mineral sands handling	183 [a]	0.97	0.12	0.73	0.73	0.73	0.73		0.088	0.011	0.066	0.066	0.066	0.066	-
Titanium-pigment	50	0.001	0.001	0.001	6.2	0.001	0.001		0.002	0.003	0.002	0.002	0.003	0.002	-
Gas-fired power plant (400 MW e)	600 [b]				230				-	-	-	-	-	-	-
Oil extraction	3 500				540					217	174	174	174	174	
Gas extraction	72 000 [b]				500					2.7	32	32	32	32	

a Zirconium.
b 10^6 m^3 a^{-1}.

Table 29
Maximum effective doses from natural radionuclides released from typical installations or operations of the mineral processing industry
[L18]

Industry	*Maximum effective dose rate (μSv a^{-1})*		
	External irradiation	*Air dispersion pathways*	*Water dispersion pathways*
Elementary phosphorus production	130	2	<0.4
Phosphoric acid production	8	~2 000 [b]	2
Fertilizer production	20	<0.4	15
Primary iron and steel production	8	<0.4	3
Coal tar processing	4	<0.4	
Cokes production	4	<0.4	
Coal-fired power plant	12	<0.4	4
Gas-fired power plant	<0.4	<0.4	–
Oil and gas extraction	2 [a]	<0.4 [b]	
Cement production	5	<0.4	
Ceramic industry plant	<0.4	<0.4	
Mineral sands handling	60	<0.4	320
Titanium pigment production	<0.4	<0.4	1

a Inhalation dose (radon) due to land fill with harbour sludge below a residential area.
b Rather uncertain value.

Table 30
Distribution of population with respect to total exposure to natural sources

Region / country	*Population (10^3) with various levels of total exposure*										
	< 1.5 mSv a^{-1}	*1.5-1.99 mSv a^{-1}*	*2.0-2.99 mSv a^{-1}*	*3.0-3.99 mSv a^{-1}*	*4.0-4.99 mSv a^{-1}*	*5.0-5.99 mSv a^{-1}*	*6.0-6.99 mSv a^{-1}*	*7.0-7.99 mSv a^{-1}*	*8.0-8.99 mSv a^{-1}*	*9.0-9.99 mSv a^{-1}*	*>10 mSv a^{-1}*
East Asia											
China (Hong Kong SAR)		1 130	4 370	770	160	46	11	5	5		
Japan	60 211	63 306	1 247								
Malaysia	12 490	4 240									
North Europe											
Denmark		800	2 800	900	400	200	50	30	30	20	20
Finland	223	1 376	2 039	687	310	154	87	82	37	29	123
Lithuania	1 680	854	770	275	80	14	28	19	5	5	
West Europe											
Belgium	280	3 300	4 500	1400	440	150	70	30	14	7	29
Netherlands	14 023	779	701	39	39						
East Europe											
Bulgaria		990	6 051	1 836							
Hungary	560	2 101	3 325	1 683	1 010	633	388	184	102	61	153
Romania		4 653	8 717	5 312	2 951	567	500				
Russian Federation	80 941	32 000	20 027	6 642	3 067	1 685	1 029	675	465	333	1 236
South Europe											
Albania	50	200	2 500	300	300	100	50				
Italy	150	15 125	25 800	7 825	4 175	2 175	1 025	500	150	150	200
Portugal	3 650	2 076	1 994	792	770	113	39				
Total	174 258	131 800	80 471	27 691	13 542	5 791	3 266	1 520	803	605	1 761
Fraction of total	0.39	0.30	0.18	0.063	0.031	0.013	0.007	0.003	0.002	0.001	0.004
Cumulative total	174 258	306 058	386 529	414 220	427 762	433 553	436 819	438 339	439 142	439 747	441 508
Cumulative fraction	0.39	0.69	0.88	0.94	0.97	0.982	0.989	0.993	0.995	0.996	1.0

Table 31
Average worldwide exposure to natural radiation sources

Source of exposure	*Annual effective dose (mSv)*	
	Average	*Typical range*
Cosmic radiation		
Directly ionizing and photon component	0.28 (0.30) [a]	
Neutron component	0.10 (0.08)	
Cosmogenic radionuclides	0.01 (0.01)	
Total cosmic and cosmogenic	0.39	0.3–1.0 [b]
External terrestrial radiation		
Outdoors	0.07 (0.07)	
Indoors	0.41 (0.39)	
Total external terrestrial radiation	0.48	0.3–0.6 [c]
Inhalation exposure		
Uranium and thorium series	0.006 (0.01)	
Radon (^{222}Rn)	1.15 (1.2)	
Thoron (^{220}Rn)	0.10 (0.07)	
Total inhalation exposure	1.26	0.2–10 [d]
Ingestion exposure		
^{40}K	0.17 (0.17)	
Uranium and thorium series	0.12 (0.06)	
Total ingestion exposure	0.29	0.2–0.8 [e]
Total	2.4	1–10

a Result of previous assessment [U3] in parentheses.
b Range from sea level to high ground elevation.
c Depending on radionuclide composition of soil and building materials.
d Depending on indoor accumulation of radon gas.
e Depending on radionuclide composition of foods and drinking water.

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PART A

Responses to UNSCEAR Survey of Exposures from natural radiation sources

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