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AIRBORNE RADIONUCLIDES AND RADIATION IN BUILDINGS: A REVIEW

A.V. Nero

August 1981

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Airborne Radionuclides and Radiation in Buildings: A Review

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ABSTRACT

This paper reviews the literature on sources and measurement of natural airborne radionuclides and radiation in buildings. It also briefly reviews control measures and suggests areas for further research. The major emphasis is given to radon 222 and its daughters, since they typically cause the largest organ dose to the general population, most of which arises from indoor exposures. The indoor radiation field from radionuclides fixed in building materials and soil is also given substantial treatment.

Keywords: airborne radionuclides, control techniques, indoor radiation, radon, radon daughters, radon sources, radionuclide contents

1. INTRODUCTION

Radiation occurs naturally throughout the biosphere, both because of primordial radioactive elements and their decay products in the earth and because of natural processes, primarily cosmic radiation, that produce either radionuclides or direct radiation fields. These natural sources expose humans to radiation both outdoors and in buildings. The purpose of this paper is to review information on airborne radionuclides

and radiation in buildings, giving particular attention to radon and its daughters, the concentrations of which are most strongly affected by building design.

In the discussion that follows, we shall refer to radionuclide concentrations and radiation fields and, by inference, to radiation doses from sources that are internal and external to the body. Radioactivity may be given in curies; 1 curie = 3.7×10^{10} becquerel, so that 1 pCi = 0.037 Bq. However, the concentration of radon daughters in air is often given as "potential alpha energy concentration" (PAEC), as discussed below. Radiation fields can be specified in terms of particle/photon or energy flux, but it is more conventional in the present context to use units of dose rate, in which case the type of radiation has to be indicated. We shall use rad as the unit of (absorbed) dose when specifying gamma radiation fields (1 rad=0.01 J/kg, so that 1 mrad=1×10^{-5}J/kg). For gamma doses, the dose in rad is numerically equal to the dose equivalent, given in rem. Note that a distinction must be drawn between the "tissue" dose, that actually received by tissue and which therefore includes self-shielding by the body, and the "air" dose, that deposited in air in the space under consideration.

The magnitude of various contributions to radiation dose vary from place to place and from outdoors to indoors, and the nature of the radiation dose depends on the radiation source. At one extreme, the cosmic radiation field delivers a dose to the entire body, but this dose is not affected greatly by the presence of a building and may be characterized primarily on the basis of altitude. At the other extreme, airborne radionuclides may cause doses specifically to the lungs, and their con-

centration indoors may be strongly affected by the nature of building materials and other sources and by building operational features that clear the indoor air. As an intermediate case, the gamma radiation field arising from radionuclides that are fixed in place typically exposes the whole body and is affected by radionuclide concentration, proximity, and shielding.

As one element in specifying the indoor radiation environment, it is useful to summarize, based on previous reviews, the dose rate contribution from natural radiation. Two recent summaries are the United Nations "UNSCEAR" report (Un77) and - for the United States - a report of the National Council on Radiation Protection and Measurements (Na75); both depend heavily on Oakley (Oa72) for U.S. data. Table 1, based on Un77, indicates doses typical of normal areas. External radiation, that arising from sources external to the body, falls into two categories, cosmic and terrestrial. The average tissue dose rate outdoors from cosmic radiation is approximately 28 mrad/y; the dose rate indoors is slightly reduced by overhead shielding (Na75 assumes 10 percent reduction of average exposures). This contribution has a significant altitude dependence, increasing from about 26 mrad/y at sea level to about 50 mrad/y at 1600 m, the altitude of Denver. The average outdoor population-weighted tissue dose rate from terrestrial radionuclides, principally due to gamma rays from ^{40}K, the ^{232}Th series, and the ^{238}U series, is approximately 32 mrad/y. This dose rate varies substantially because of geographic variations in the distribution of these radionuclides. In estimating average terrestrial dose rates, Un77 used an indoor tissue dose rate approximately equal to the outdoor rate; in contrast, Na75 assumed - for U.S. housing - that indoor dose rates were 20

percent less than outdoor rates. Internal radionuclides contribute significant beta and gamma doses to much of the body (about 20 mrad/y, primarily from ^{40}K) and a significant alpha dose to specific organs (even excluding that to the lungs from radon and its daughters). The alpha dose arises primarily from internally deposited $^{238,234}U$, $^{226,228}Ra$, and ^{210}Po and varies greatly with body organ. One of the larger contributions, about 3 mrad/y, is the ^{210}Po alpha dose to the cells lining the bone surfaces. Alpha particles have a greater biological effectiveness than gamma rays, so that the alpha absorbed dose contributes a dose equivalent an order of magnitude greater than that of the same (absorbed) dose of gamma radiation. Table 1 shows, for various organs, estimated dose equivalent rates, in mrem/y, which are numerically equal to tissue dose rates (in mrad/y) for gamma and beta radiation. For calculating the dose equivalent from alpha radiation, a quality factor of 20 was assumed (based on relative biological effectiveness), in accordance with recent recommendations (In77). The value given for lung dose from inhaled radionuclides assumed a ^{222}Rn concentration in air of 1 pCi/l (and half an equilibrium amount of its daughters). The resulting dose equivalent (600 mrem/y) dominates the dose equivalent to the lung, which has the largest value in the table. Even within the lung, the dose from radon daughters varies substantially, with the largest energy deposition in tissues of the tracheobronchial tree.

Although all indoor dose rates from natural radiation sources are affected by buildings, those from inhaled radionuclides are affected most strongly. The only natural airborne radionuclides of significance are radon and its daughters, principally the series beginning with ^{222}Rn, the alpha decay product of ^{226}Ra (a member of the ^{238}U series).

Radon is a noble gas that can move from the site of its formation, giving it significant opportunity for reaching air that is inhaled by humans. The short-lived decay products of radon, i.e., polonium, lead, and bismuth, are chemically active and thus can be collected in the lungs, either directly or via particles to which they attach. The most significant dose arises from alpha decay of the polonium isotopes. The decay sequence beginning with ^{226}Ra is shown in Figure 1, and, from the biomedical point of view, effectively ends with ^{210}Pb, because of its half-life of 22 years. Because the alpha energy associated with decays of the short-lived daughters to ^{210}Pb poses the main risk, daughter concentrations are often expressed as the associated "potential alpha energy concentration" (PAEC) of the short-lived daughters in air. The unit conventionally used for PAEC is the working level (WL), defined as 1.3×10^5 MeV/l, the PAEC if approximately 100 pCi/l of ^{222}Rn were present with equilibrium amounts of its daughters. Dose (and dose equivalent) rates may be estimated from the PAEC on the basis of relatively complicated modeling, provided that the daughter particle size distribution and other factors are prescribed.

The character of a building may affect occupant radiation exposure in three principal ways: (1) the building serves as a container for indoor-generated radon and its associated daughters, whether from building materials, underlying soil, or water and gas; (2) building materials may contain natural gamma emitters (^{40}K, ^{232}Th series, ^{238}U series); and (3) the building shields occupants from external cosmic or terrestrial radiation. The last two effects tend to cancel one another. The building structure may, in unusual circumstances, also protect occupants from outdoor radon daughter concentrations. However, the indoor

concentration is ordinarily larger than the outdoor value, and outdoor-generated radon usually contributes a small additive term to indoor concentrations. Ignoring this term, the steady-state indoor radon concentration for a fixed indoor radon source strength is inversely proportional to the air exchange rate, the rate at which the indoor air is exchanged for outdoor air. Air exchange rates for most U.S. buildings are on the order of 1/h, with 0.5 to 1.5/h typical for residences (windows closed), but lower rates are desirable for reducing energy use and are in fact typical in many other countries. The air exchange rate and other removal mechanisms also affect the ratios of the radon daughter concentrations to the radon concentration. Activity ratios of one could only occur if no removal mechanisms were active and, as expected, substantially lower values have been observed. An equilibrium factor (F) is often defined as the ratio of the actual PAEC to the PAEC that would be associated with the actual radon concentration if the daughters were in equilibrium with this radon.

Possible sources of radon in buildings include building materials, the soil and rock underlying structures, and water or gas supplied to the buildings. In many cases, such as single-family residences in the United States, the underlying natural materials appear to constitute the principal source of radon. In larger structures, the building materials may contribute a greater share of the source strength, but the absolute contribution is usually small for most materials. However, certain materials have been found to constitute unusually large sources of radon and even of gamma radiation, a notable example being the alum-shale concrete present in ten percent of Swedish houses. Largely because of such cases, at least one international entity has recently reviewed the

contribution of building materials to radiation exposures (Nu79).

This paper characterizes sources of indoor airborne radionuclides and radiation, summarizes measurements of actual concentrations or radiation fields, briefly indicates control measures, and suggests areas for further research. The major emphasis is given to radon and its daughters, since they typically cause the single largest organ dose, which arises primarily from indoor exposures. However, the indoor radiation field from radionuclides fixed in building materials and soil is also given substantial treatment. This radiation arises principally from several primordial radionuclides with concentrations on the order of 0.1 pCi/g or greater in rocks, soil, and derivative building materials, i.e., ^{40}K and members of the ^{232}Th and ^{238}U decay series. These are also the decay chains in which ^{220}Rn, ^{222}Rn, and their daughters occur.

2. SOURCES OF RADIONUCLIDES AND RADIATION

a. Building materials

Radionuclide content

Surveys of the radionuclide content of building materials in Europe are summarized in Un77 (p. 50, Table 8), which gives activity concentrations of ^{40}K, ^{226}Ra, and ^{232}Th. Average values for the concrete sample groups examined range from 0.9 to 2.0 pCi/g for ^{226}Ra, 0.8 to 2.3 pCi/g for ^{232}Th, and 9 to 19 pCi/g for ^{40}K. By comparison, the ranges for brick are about 50 percent higher; those for cement are similar except for ^{40}K (which is 50 percent less); and those for natural plaster are lower by about a factor of five. Recent measurements (e.g., St80b) give

similar results. Concentrations for building materials not derived from crustal components, e.g. wood, are much lower.

Recent U.S. data (Table 2) show concentrations in the same range, assuming that the series radionuclides are sufficiently close to equilibrium to permit comparison. Radium 226, in particular, is part of the ^{238}U series. In a number of cases, U.S. workers have examined the radionuclide contents of concrete in the course of selecting materials for low-background radiation-counting facilities (Ll79); the values obtained are consistent with the European data, although somewhat lower. The observed concentrations are also within the range of values typical for major rock types and soils (see below). Measurement programs have recently been initiated to characterize the radionuclide contents of U.S. building materials as a basis for understanding the resulting effect on the indoor radiation environment. Kahn et al (Ei78, Ka79) have reported measurements of concentrations in various building materials in the Atlanta area. Ingersoll (In81) surveyed concretes and other materials as part of a program on indoor air quality; radionuclide contents for concrete and rock bed samples from a number of areas are given in Table 2.

Certain natural materials have higher radionuclide concentration than noted for ordinary concretes. Granite often exceeds the ranges given in Table 2. Shales in the southeastern United States are known to exceed average crustal ^{238}U concentrations by an order of magnitude. A substantial fraction of Swedish houses built during the period 1930-1975 used aerated concrete incoporating alum shale and having ^{226}Ra concentrations in the vicinity of 40 pCi/g (Sw78).

Considerably greater radionuclide concentrations than given in Table 2 may also be found in building materials using residues from industrial processes. Nu79 reviews use of such materials, examples being the Federal Republic of Germany's use of "red mud" (from bauxite processing) for bricks and of blast-furnace slag for blocks. A possibility with wider implications, as noted in Nu79, is use of wastes from processing sedimentary phosphate ore, which contain substantial concentrations of the ^{238}U series. Phosphogypsum, essentially calcium sulfate produced by treatment of phosphate ores with sulfuric acid, may be used for building materials, particularly wallboard. In this treatment ^{226}Ra follows the calcium, leading to tens of pCi/g in the gypsum. Although the United States is the largest producer of phosphate rock, use of phosphogypsum is most attractive for other countries having little natural gypsum (Nu79). The principal example of byproduct use in the United States is concrete blocks incorporating phosphate slag (essentially calcium silicate), which contains most of the ^{226}Ra and ^{238}U contained in the phosphate ore (Ro79). For the electric furnace process used in Florida, concentrations in the ore are in the vicinity of 60 pCi/g, and the slag has similar concentrations. Until 1978, a plant in Alabama (using Florida and Tennessee phosphate ores) sold slag to companies in Alabama, Mississippi, Tennessee, Georgia, and Kentucky. The concrete produced by these companies has ^{226}Ra concentrations estimated (and, in some cases, measured) to be about 20 pCi/g and may have been used in approximately 100,000 homes (Ka79). As a final example of potential importance for the United States, some fly ash from coal-fired power plants has been used in cement production, and this use may continue. Heretofore it has not been thought to contribute significantly to the radionuclide content

of the resulting building material (Ka79). Emanation measurements on fly-ash concretes are now being performed (Sm81).

Radon emanation

The effective ^{222}Rn generation rate in building materials depends on the ^{226}Ra content, which varies widely as discussed above, and on the percentage of radon formed that does not remain lodged in the matrix of the material. Radon that is not fixed in place may move through the matrix by diffusion or, if large air spaces exist in the material, by convection. Diffusive movement depends on the diffusion length of the material in question and on the material's thickness. (Diffusion of radon in concrete has been measured by Kr71, Cu76a, Jo78, St80b, and Za81.) The extent to which other transport processes occur depends not only on the material's characteristics, but also on environmental conditions, i.e., pressure, temperature, and moisture content. A rule of thumb sometimes cited (e.g., Un77, p. 75) is that one percent of the ^{222}Rn generated from materials in walls and ceilings escapes into the adjacent airspace. However, recent measurements have indicated that a considerably higher fraction can escape: Stranden (St80b) observed escape-to-product ratios up to 20 percent for concrete; Ingersoll (In81) cites escape-to-production ratios of 8-25 percent for the concrete sample groups indicated in Table 2.

Of most direct interest for indoor air quality is the actual emanation rate, often given as pCi $m^{-2}s^{-1}$ and sometimes as pCi $g^{-1}s^{-1}$. Measurements for various materials give emanation rates over a wide range. For example, European gypsum board and bricks yield, respectively, about 0.3×10^{-4} and 1×10^{-4} pCi $m^{-2}s^{-1}$ of ^{222}Rn, while rates for European

concretes range from 0.001 to 0.2 pCi $m^{-2}s^{-1}$ (Jo78). Measurement of ^{222}Rn emanation rate per unit mass for sample groups of concrete from U.S. metropolitan areas (Table 2) give averages that range from 0.4 to 1.2 pCi $kg^{-1}h^{-1}$; 0.8 pCi $kg^{-1}h^{-1}$ yields approximately 0.03 pCi $m^{-2}s^{-1}$ for 0.1 m thick concrete. Several rock samples from solar storage beds averaged 0.5 pCi $kg^{-1}h^{-1}$, although (as indicated in Table 2) ^{226}Ra contents were considerably higher than those for the concrete samples (In81).

The resulting indoor ^{222}Rn concentrations depend on the amount of such material in the structure, the interior volume, and the air exchange rate. For an air exchange rate of 1.0 h^{-1} and an indoor emanating surface to indoor volume ratio of 0.5 m^2 per m^3, an emanation rate of 0.03 pCi $m^{-2}s^{-1}$ corresponds to an ^{222}Rn concentration of about 0.04 pCi/l. Depending on the equilibrium factor, this would yield a PAEC of about 0.0002 WL. On the other hand, the much higher radionuclide contents of Swedish alum-shale concrete yields much higher airborne concentrations, as discussed below. Direct measurement of emanation rates of materials made with industrial by-products (such as phosphate slag concrete) is underway. Considering that these materials may contain 100 times as much ^{226}Ra as the average for concrete, contributions of up to several pCi/l of ^{222}Rn and a corresponding increase in the PAEC could be expected if the same emanation ratio pertained, but preliminary results suggests a lower ratio (Sm81).

In some cases, such as Swedish houses using alum-shale concrete, building materials are known to contribute substantially to indoor radon concentrations. Such cases are to be expected considering the wide

range in radionuclide concentrations that occur in building materials. However, because emanation rate per unit of activity concentration varies by more than an order of magnitude (Un77), it is difficult to use radium content to predict the contribution of a particular material to indoor radon levels. For this reason, more comprehensive information on diffusible fraction, diffusion length, etc., and their dependence on material or environmental factors is required before we can characterize building materials on the basis of radionuclide content. Should this information become available, radionuclide contents may then be helpful in characterizing indoor levels on a broad scale, e.g., by geographic area. However, the fact that diffusion and emanation rates also depend on environmental factors, such as pressure and temperature, and on the moisture content of the material may limit the possibility for such characterization.

In some cases, ^{220}Rn ("thoron") and its daughters, ordinarily present at much lower concentrations than ^{222}Rn and its daughters, may assume importance, particularly when mechanisms exist for transporting emanating ^{220}Rn rapidly into the airspace of interest. In comparison with ^{222}Rn, the much shorter half-life of ^{220}Rn, 55 s, causes the measured activity concentration to be a parameter of secondary interest. However, the PAEC still gives a useful indication of possible dose to the lung, assuming that a realistic model is used (see, for example, Ja72b). One WL of ^{222}Rn daughters has the same PAEC as that associated with daughters in equilibrium with 7 pCi/l of ^{220}Rn. To the extent that ^{238}U and ^{232}Th, which have similar half-lives, have similar activities in source materials (as is typical in Table 2), the PAEC from their progeny, the ^{220}Rn and ^{222}Rn daughters, can reach similar values if rapid

transport mechanisms exist. This may occur, for example, in solar buildings that sweep air through rock or concrete thermal storage beds. Techniques have recently been developed to measure ^{220}Rn emanation rates (Mc180).

Gamma Radiation

The energies and intensities of photons from decay of natural radionuclides have been well characterized. The external dose from radionuclides in building materials arises from the gamma rays emitted, and depends on the geometry of the structure and attenuation by the materials, as well as on the gamma-ray energies. A simple expression may be derived for the gamma air dose rate (in μrad/h) in a hole in an infinite uniform medium (Ka79):

$$\dot{X}_{\infty} = 2.43\ (E_U\ C_U + E_{Th}\ C_{Th} + E_K\ C_K),$$

where C_U, C_{Th}, and C_K are the concentrations (in pCi/g) of ^{238}U and each of its daughters, ^{232}Th and each of its daughters, and ^{40}K, respectively, and E_U, E_{Th}, and E_K are the average gamma-ray energy (in MeV) per disintegration of the indicated radionuclide (including disintegration of the daughters, for the ^{238}U and ^{232}Th series). Using E_U = 1.72 MeV, E_{Th} = 2.36 MeV, and E_K = 0.156 MeV (Ka79), $\dot{X}_{\infty}$ = 4.2 C_U + 5.7 C_{Th} + 0.38 C_K in μrad/h. The stated dose contributions from the uranium and thorium series are slightly less than those cited elsewhere, e.g., by Krisiuk et al (Kr71), who may have used older information on decay schemes. For the radionuclide contents cited in Table 2, the three terms in the expression for $\dot{X}_{\infty}$ contribute comparable amounts. (An analogous expression for the dose from a flat plain is cited in the

section on soil.)

For an actual structure, the geometry is complex and varied; in addition, the building materials may attenuate the external radiation dose from other sources. Moreover, ^{222}Rn and its daughters may be present in the material at less than equilibrium values, thereby decreasing the corresponding gamma dose. The ^{222}Rn escape-to-production ratio is most often in the few to 25 percent range, causing only a small reduction in the value of $\dot{X}$. The effects of geometry and attenuation cannot be so simply characterized. Dose rate expressions from various workers, pertaining to a variety of structures, have been summarized (Nu79). Some of these expressions account for reduction of the dose rate from outdoor sources. Moeller et al (Mo78) describe a computer program suitable for analysis of varied geometries.

The infinite geometry case yields air dose rates in the vicinity of 8 μrad/h for a ^{40}K concentration of 8 pCi/g and $^{238}U/^{232}Th$- series radionuclide concentrations of 0.5 pCi/g. A thick slab of such material would contribute about half this dose rate at its surface. As discussed earlier, a typical outdoor tissue dose rate from terrestrial radionuclides is 32 mrad/y or 4 μrad/h.

b. Soil

Radionuclide content

Radionuclide concentrations for major rock types and soil have been summarized. The UNSCEAR report (Un77) cites world-average soil values of 0.7, 0.7, and 10 pCi/g for ^{238}U, ^{232}Th, and ^{40}K, respectively, with a typical range of roughly a factor of six in each case. These average

values are similar to those based on 200 measurements of gamma dose rate cited by Lowder et al (Lo64). Values for crustal rocks (Na75, Un77) typically lie in the vicinity of these concentrations, but certain formations have considerably elevated values. For example, the phosphate rocks of Florida contain the ^{238}U series at tens of pCi/g, but with normal amounts of ^{232}Th; commercial uranium ore bodies in the U.S. and elsewhere have ^{238}U concentrations of hundreds of pCi/g and higher; shales in the Tennessee area have ^{238}U concentrations of tens to hundreds of pCi/g.

Radon emanation and transport

The ^{238}U series, typically present in soils and rocks at concentrations of about 1 pCi/g, includes ^{226}Ra, the source of ^{222}Rn. The actual ^{222}Rn emanation rate from the ground depends, as for building materials, on the percentage of diffusible radon, diffusion length, and other transport mechanisms (including groundwater) in the soil. A review of available measurements of ^{222}Rn indicates a mean emanation rate from the soil of 0.42 pCi $m^{-2}s^{-1}$ (Wi72). Given this value for the ground under a one-story house, and assuming the radon emanated finds its way into the indoor air, the soil could account for indoor ^{222}Rn levels of about 1 pCi/l at typical air exchange rates of 1 h^{-1}. Because emanation rates range over at least an order of magnitude from place to place, this potential contribution can also be expected to vary substantially within buildings. Considering that building materials ordinarily do not contribute as much ^{222}Rn, multistory buildings can be expected to have lower concentrations than single-story buildings in most cases.

The soil as a source of ^{222}Rn could be characterized directly by emanation measurements or, alternatively, if disequilibrium and transport mechanisms (including groundwater) were known, indirectly by measurements of members of the ^{238}U series. Because of the relative ease of measuring gamma rays, the indirect method may be more appropriate for large-scale surveys intended to characterize the contribution of soil radon by geographic area. Gamma-ray source measurements may also be less sensitive to changes in pressure, temperature, and moisture content than emanation-rate measurements (Un77; Na75). A further consideration is that variations in emanation rate may correlate with factors that affect air exchange rates and, as a result, may complicate assessment of the importance of soil as a source of indoor radon.

The mechanisms by which radon may be transported into buildings have been given little study. Soil-gas measurements, which have yielded results from 100 to 2000 pCi/l (Kr64; Sc78), may be relevant to this question since they may help in characterizing the radon content of air trapped beneath building foundations. Emanation rates per se are useful only for placing an upper limit on the potential of soil as an indoor source. However, a more detailed understanding of the way in which radon is transported in soil could provide a basis for using emanation data to estimate the amount of radon that may accumulate beneath houses and be transported indoors. Such collection and transport mechanisms may be strongly affected by changes in barometric pressure, soil moisture content, temperature gradients, and wind.

The actual pathway by which radon enters a building from the soil appears to vary substantially with building design and construction

practice. In houses with concrete basements that are closed to the outdoors, radon may enter by diffusion through the basement floor, by convection within basement walls, and by movement through cracks and designed openings or penetrations in either of these components. Even in communities where numerous measurements have been performed, it has not been possible to determine the relative importance of these mechanisms. In some mining communities, sealing of cracks has proved relatively successful in reducing radon levels (At78, At79, At80), but the effectiveness of this method in general has not been evaluated. Even as basic a question as the effectiveness of concrete itself as a barrier against radon entry has not been answered, although information on diffusion coefficients is improving (Kr71, Cu75, Jo78, St80b, Za81).

The subsequent movement of radon from the point of entry to other parts of the building depends on internal construction and building use. Even in buildings with ventilated crawlspaces, the radon concentration in the crawlspace air may still be considerably higher than outdoors, and a significant amount of the radon emanating from the soil may reach the interior space by transport from the crawlspace.

More comprehensive information on how radon is transported is needed in order to develop techniques for preventing radon from entering buildings. It is also needed so that a correlation can be established between the ^{226}Ra content in soil and indoor ^{222}Rn levels attributable to this source.

Gamma Radiation

The gamma dose from radionuclides in soil may be expressed in a fashion analogous to that for building materials; the air dose rate (μrad/h) at one meter above the ground due to natural emitters uniformly distributed in the soil (and with the ^{238}U and ^{232}Th series in equilibrium), has been given as $\dot{X}_{plane} = 1.82\ C_U + 2.82\ C_{Th} + 0.179\ C_K$ for C_U, C_{Th}, C_K in pCi/g (Be72). More current data on decay schemes may alter this slightly. As noted above, concentrations of natural radionuclides in soil and rock vary from place to place, causing comparable variations in dose rates. As examples, the air dose rate is estimated at 2.6 μrad/h on the U.S. coastal plain (the Atlantic and Gulf coastal areas), 10.2 μrad/h on the Colorado Plateau, and 5.2 μrad/h on the rest of the contiguous United States (Na75), based on nuclear plant site surveys. Un77 adopts a world land average of 4.6 μrad/h. The materials in a building can provide significant shielding of occupants from gamma rays from local radionuclide concentrations, but the radionuclide content of the materials may more than compensate for this shielding (see below).

c. Radon from utilities

Water

Measured concentrations of ^{222}Rn in well water in Maine and New Hampshire average 53,000 pCi/l and 101,000 pCi/l, respectively (Lu64). More recent measurements have been performed in Maine (He78; Ge80). Snihs (Sn73) reports concentrations up to 50,000 pCi/l in Swedish drilled wells. Deep wells in Finland are widespread enough and have sufficiently high radon concentrations (averaging 17,000 pCi/l, accord-

ing to As80) to give rise to a population of thousands receiving radon-daughter exposures exceeding occupational limits (Ca80). Concentrations of 100-7500 pCi/l have been found in tap water from wells or underground reservoirs associated with U.S. houses in which indoor concentrations were measured (Ho80). ^{222}Rn in water can quickly transfer to air, with efficiencies of 30-90 percent, depending on water use (Ges78, Pa79); a concentration of 10,000 pCi/l can raise average indoor ^{222}Rn levels by on the order of 1 pCi/l. It is not known generally how widespread such high concentrations in water are, nor how closely they correlate with high radium content in surface soils and rocks.

Natural gas

Concentrations of ^{222}Rn in natural gas in the Houston area have been found to average approximately 50 pCi/l at STP (Ge73). Concentrations in distribution lines at various points in the United States were found to average about 20 pCi/l (Jo73). The resulting levels in U.S. residences due to natural gas combustion has been estimated to be less than 0.1 pCi/l, even with unvented burners.

3. INDOOR CONCENTRATIONS AND RADIATION FLUXES

a. Airborne radionuclides

Radon concentrations

Data from studies surveyed in the UNSCEAR report (Un77) indicate that indoor ^{222}Rn concentrations vary by two orders of magnitude, with average values in the vicinity of 1 pCi/l. Such a large range is not surprising considering that the studies included various types of build-

ings, building materials, underlying materials, and ventilation conditions and used many different measurement techniques. More recent measurements, many of which were reported in two recent conferences (Na78, Sp80), have confirmed this wide variation (see below).

Such a wide variation is expected, even for conventional housing, since the ventilation rate varies over a wide range. Moreover, depending on the location and nature of a building, the radon source strength can vary substantially. The soil underlying a single-family house can ordinarily be expected to be a principal contributor to its indoor radon concentration. As noted earlier, a typical soil emanation rate, if injected into the interior of a house with an air exchange rate of $1h^{-1}$, would contribute about 1 pCi/l of ^{222}Rn. Since soil emanation rates and effective capture by the house vary by three orders of magnitude (Ne81), and air exchange rates vary significantly, a large range in indoor concentrations would result. Similarly, construction materials and ventilation rates directly affect the radon concentrations of larger buildings.

As indicated in Table 3, homes monitored in New York and New Jersey were found to have an annual average ^{222}Rn concentration of 0.3-3.1 pCi/l in the living space, with a geometric mean of about 0.8 pCi/l (Geo78). Similar measurements in Austria yielded an arithmetic mean of 0.6 pCi/l (St78) and, as is characteristic of results approaching a log-normal distribution, a somewhat lower geometric mean. In these studies the mean indoor concentrations were three to four times as great as local outdoor concentrations. Mean indoor concentrations of radon in the Oslo area were found to be 1.3, 2.0, and 1.0 pCi/l for buildings of

wood, concrete, and brick, respectively (St79). Grab sample measurements of homes in the San Francisco area made during the summer with windows closed and with an average air exchange rate of 0.4 h^{-1}, gave concentrations averaging 0.3 pCi/l (Be79); considerably higher concentrations were found in a rural part of Maryland (Mo81). Grab sample measurements in Illinois showed a substantial incidence of concentrations greater than 5 pCi/l; 6 of 22 houses had concentrations of 10 pCi/l or more (Ru79). Unpublished data (Pf81) indicate high average concentrations in a part of eastern Pennsylvania. A survey of 10,000 Canadian homes, in which measurements were taken primarily in basements, gave geometric mean concentrations ranging from 0.14 to 0.88 pCi/l for the 14 cities monitored (Mcg80).

High ^{222}Rn concentrations have been found in uranium mining areas and in buildings that use materials high in radium content. In houses monitored in Bancroft, Ontario, 50 percent of the sample had concentrations greater than 3 pCi/l, over 25 percent had concentrations greater than 7 pCi/l, and about 6 percent had concentrations greater than 15 pCi/l (Ma79). High levels have also been found in homes in mining areas in the United States; at Grand Junction, Colorado, PAECs corresponding to ^{222}Rn concentrations up to hundreds of pCi/l have been measured. In a survey of several Swedish houses built with alum-shale-based concrete, the average ^{222}Rn concentration was 7 pCi/l (Sw78); more recent data give average concentrations of 15 pCi/l or more for residences built entirely of such concrete (Sw80).

Concentrations of ^{222}Rn of 0.6-22 pCi/l have been found during grab-sample measurements of energy-efficient homes in the United States,

many of which had low air exchange rates; these measurements were taken with windows closed, and the air exchange rates were measured simultaneously (Ho80). Concentrations and air exchange rates have also been measured in conventional houses in England (Cl78) and in houses at Elliot Lake, Ontario (Sm79); the measured ^{222}Rn concentrations were consistent with those observed for conventional houses elsewhere (see above). From his data, Cliff (Cl78) inferred a large range in ^{222}Rn source magnitude (a median of 0.32 pCi $l^{-1}h^{-1}$, with a geometric standard deviation of 3.1), consistent with comparable extraction of source magnitudes from U.S. data (Ne81). Winter measurements in New York yielded average ^{222}Rn concentrations of 1.0 pCi/l for conventional houses and 6.4 pCi/l for "energy-efficient" houses (Fl81); ventilation rates were not measured, and most of the "energy-efficient" average is contributed by a single solar home that, apparently because of a particular heat storage medium, had a large source magnitude.

The range in observed indoor concentrations arises, not only from differences from one house to another, but also from temporal variations in source magnitude or ventilation rate in individual houses. Spitz et al (Sp78) have examined the temporal variation of indoor ^{222}Rn concentrations in houses in Colorado and New Jersey and the influence of ventilation conditions. Steinhausler (St75) has examined the correlation between meteorological variables and indoor concentrations. In a review of outdoor and indoor ^{222}Rn concentrations, Gesall examines variation with time and place (Ge81).

Radon daughter concentrations and behavior

Radon daughter concentrations are often measured as potential alpha-energy concentrations (PAEC), given in working level (WL). UNSCEAR (Un77) reviewed pre-1977 measurements, and numerous more recent papers appear in Na78 and Sp80. For the New York and New Jersey houses, referred to above, the annual-average PAEC for ^{222}Rn daughters had a geometric mean of about 0.004 WL in the living space, with a range of values from one house to another of 0.002-0.013 WL; equilibrium factors averaged slightly above 0.6 in the living space (Geo78). Stranden et al (St79) found mean equilibrium factors of 0.5 in dwellings in the Oslo area. Equilibrium factors of about 0.4 have been found in Sweden (Sw78). Based on earlier work, Un77 adopted a typical value of 0.5 for its estimates of exposures.

Measurements in Florida houses built on reclaimed phosphate land yielded average ^{222}Rn daughter concentrations in the vicinity of 0.01 WL, but the range extended above 0.05 WL (Gu78). Houses in Grand Junction (Colorado) in which remedial action has been recommended had PAECs ranging from 0.02 to 1 WL. Sets of control houses monitored in Florida and Colorado had an average PAEC similar to that in New York and New Jersey (see table 3). Measurements have also been performed in homes in the vicinity of uranium mining operations (e.g., At78, At79, At80).

In some cases, concentrations of individual ^{222}Rn daughters have been measured. Based on work cited in Un77, typical activity ratios for $^{222}Rn/^{218}Po/^{214}Pb/^{214}Bi$ in residences are 1.0/1.0/0.6/0.4, but with significant variability among surveys. A survey of hundreds of residences in Hungary gave higher daughter ratios (averaging 1.0/0.9/0.8) and mean

^{222}Rn concentrations ranging from 0.7 to 5.8 pCi/l for various housing types (To72). More recent measurements of individual daughter concentrations, typically yielding lower ratios, have often been made in order to correlate concentrations with possible removal processes; see discussion of potential control techniques below.

Some work has been done on characterizing the distribution of particle sizes of indoor ^{222}Rn daughters as well as the dependencies of concentrations and distributions on various parameters, including location, particulate mass concentration, air exchange rate, and air-mixing rate. The fraction of ^{222}Rn daughters that is not attached to particles, as well as the size distribution of attached daughters, was measured in a laboratory building and in four homes (Ge72, Geo78). Such measurements have also been performed in uranium mines. The diffusion coefficients of radon daughters have been subjects of intensive experimental study (recent examples being Ra79, Po79, and Bu81).

The simplest models of indoor radon and radon daughter concentrations utilize mass-balance equations connecting the indoor radon source strength, outdoor concentrations, and the air exchange rate, assumed to be the only removal mechanism other than radioactive decay. (See, for example, Ku79.) Models may also simulate diffusion of radon into the house (Mo78), but transport has not been modeled in any comprehensive way. Models have also been made of radon daughter diffusion and attachment processes (Ra69; Wr69) and of the effect of such processes on daughter concentrations and unattached fractions (Ja72, Po78, Ho79). However no realistic models of radon and daughter behavior in buildings, by which actual concentrations (or the effect of control measures) might

be simulated, has been attempted. More experimental information will evidently be required to validate such models. On the other hand, attempts have been made to simulate, on a practical basis, average radon concentrations (St80b) or radon daughter exposures (Sw78, Mc81).

In some cases, daughters of radon 220 may be present in concentrations that are comparable to ^{222}Rn daughters, at least as measured in terms of potential alpha energy concentration. Stranden (St80a) found a mean ^{220}Rn daughter PAEC of 0.0025 WL in 22 dwellings in Norway, half the PAEC observed for ^{222}Rn daughters. In a few measurements of ^{220}Rn daughters in solar homes in New Mexico, PAECs were found to be approximately 0.005 WL (In80).

b. Gamma radiation fluxes and shielding effects from building materials

As discussed above, gamma-ray doses may arise from terrestrial radionuclides both in building materials and in nearby soil and rock, and the radionuclide content of these two sources may vary significantly. Moreover, the structural materials serve to shield occupants both from gamma rays from soil and rock, and, to a lesser degree, from cosmic rays. As a result, the building may affect external dose rates of occupants in various ways and degrees. Given information on a particular building, the net effect may be calculated in a way similar to the work of Moeller et al (Mo78), based on the gamma dose rate expressions given above and on estimations of shielding effects.

In some cases, the structure may have little effect on terrestrial or cosmic dose rates. Exclusive use of materials that do not contain

significant radioactivity, such as wood, has the effect of shielding the terrestrial gamma flux (about 32 mrem/y tissue dose) by about 20 or 30 percent and has little effect on the cosmic ray dose (about 28 mrem/y). A concrete foundation (slab floor or basement) has no effect on the cosmic ray dose and, if its radionuclide content is similar to that of surrounding soil or rock, little effect on the terrestrial dose. That is, although concrete substantially attenuates gammas from the soil or rock, it contributes a gamma flux that compensates for this reduction.

On the other hand, if a building also uses concrete in the walls and ceilings, and has a radionuclide content similar to that of local soil and rock, an approximate doubling of gamma dose rates from terrestrial radionuclides occurs. As some compensation, concrete walls and ceilings tend to shield occupants from cosmic rays, in many cases by only about 20 percent, but by larger factors for large buildings.

Ordinarily, then, building materials having crustal components with radionuclide contents similar to that of local soil and rock may increase external dose rates for occupants by up to tens of mrem/y or may decrease rates by a somewhat smaller amount. Table 1 presumed a higher air dose rate indoors than outdoors, corresponding to the case where the gamma dose from the building materials exceeded the reduction in dose due to shielding of gamma rays from surrounding soil and rock. This contrasts with the U.S. situation, where the net effect of building materials is thought to be a slight reduction in the total dose rate. For building materials and surrounding soil or rock that contain higher radionuclide contents, the dose rate differences between outdoors and

indoors would be correspondingly larger.

4. CONTROL TECHNIQUES

From the few available indoor measurements of ^{222}Rn daughters, it appears that variations of 0.01 WL from one building to another, depending on air exchange rates and on building or ground materials, are not unusual. The full range of values for conventional houses has a considerably larger variation than this, largely because of differences in source strength, and measures that reduce the air exchange rate substantially can be expected to change concentrations by corresponding amounts. A daughter concentration of 0.01 WL, if experienced two-thirds of the time, corresponds to an exposure of about 0.3 WLM per year, about a factor of 10 less than the occupational limit of 4 WLM/y. (Exposure of an individual to 1 WL for 170 hours, a working month, yields one "working level month" or WLM.) On the other hand, variations in external dose rate due to ordinary building materials are on the order of 10 mrem/yr, less than one-hundredth of the whole-body occupational dose limit of 5 rem/y. If these occupational limits correspond to similarly valued risks, it appears that the effect of the structure on radon daughter exposures (given in WLM/y) is far more significant than the effect on external whole-body dose rates (given in rem/y), except in cases involving materials with unusually high radionuclide concentrations. For this reason, the discussion that follows emphasizes methods for controlling radon daughter exposures. One of these methods, materials substitution, may also be used for control of gamma dose rates, particularly in cases where materials have unusually high radionuclide contents. An additional technique for control of gamma doses is installa-

tion of shielding materials that do not themselves contribute substantial radiation doses.

Techniques for control of indoor levels of ^{222}Rn or its daughters include measures that decrease ^{222}Rn sources, reduce transport from sources, remove ^{222}Rn or its daughters from indoor air, or exchange indoor air for outdoor air. The easiest technique to implement in many cases is to increase the air exchange rate, for example by opening windows or installing fans. For reasons of comfort or energy efficiency, other methods, sometimes equally straightforward, may often be preferable. In general, not enough is now known about the cost, effectiveness, and applicability of various measures to determine their role in the general building stock.

a. Material selection or site preparation

Construction of a building with materials having low ^{222}Rn emanation rates can affect the source strength directly. In some cases, attention to materials processing may reduce emanation rates. However, in situations where the surrounding soil and rock contribute most of the ^{222}Rn, opportunities for controlling the source strength are limited, especially since the diffusion length of ^{222}Rn is relatively large and radon source strength is seldom a criterion for site selection. Attention to building materials or site materials (underlying and surrounding soil) in new construction has a significant effect in cases where the emanation rate from either of these may be unusually high. Replacing such materials (on a remedial basis) is often difficult or expensive, in which case other measures may be favored. However, replacement has been a principal technique in communities associated with uranium mining or

processing (At78, At79, At80).

b. Reducing transport

The principal means of reducing the transport of radon to building interiors are to seal materials having significant emanation rates or, for the case of transport from surrounding soil, to plug cracks or holes through which air with a high ^{222}Rn content (i.e., soil gas) moves. Materials may be sealed by epoxies or other coatings with up to 90 percent effectiveness (Kr71; Cu73; Au74; Cu78). Sealing surfaces, filling holes with impervious materials, or stopping transport via installation of plastic or other barriers has proved effective in some cases that required remedial action (see, for example, At78, At79, At80), but they all require integrity of the barrier for long-term transport reduction. The general applicability or effectiveness of these measures as long-term passive controls is not known. (It should be noted that confinement of radon by diffusion or convection barriers also permits buildup of radon and its daughters behind the barrier, causing an increase in gamma irradiation from building materials. Nevertheless, this increase appears less significant than the associated decrease in airborne ^{222}Rn and daughters [Cu76b]). Transport may also be reduced by ventilating crawlspaces or basements or by designing transport routes that by-pass slab floors or basements. This has been employed in the Canadian communities with remedial action programs (At80).

c. Removal of daughters from indoor air

Methods for removing ^{222}Rn daughters from indoor air include: (1) filtration using fiber, electrostatic, or charcoal filters; (2) mixing

of indoor air to cause deposition within the structure or ventilation system; and (3) space charging to remove daughter ions. Filtration systems are effective in reducing airborne particulate mass concentrations. However, depending on the system, they may thereby raise the concentration of unattached daughter ions, especially ^{218}Po (Gu79); for some particle size distributions, this would raise the ratio of lung dose to PAEC. Nazaroff et al (Na81) observed a substantial decrease of PAEC from operation of the furnace fan (which thereby activated the system's filter), but the unattached fraction was not measured. Guimond and Windham (Gu78), Holub et al (Ho79), and Jonassen (Jo80) have performed related experiments on air mixing, ventilation, and filtration. Finally, in many measurement techniques, charged ^{222}Rn daughters are collected by voltage differentials, but it does not appear that this principle can easily be applied as a control measure.

d. Exchange of indoor and outdoor air

Use of air-to-air heat exchangers to remove indoor air while recovering potentially lost energy is now being investigated. Preliminary results (Na81) indicate that this method is effective, in at least one configuration, in reducing ^{222}Rn and daughter concentrations. This method is particularly attractive because it can be applied in both new and existing buildings and because it is effective in reducing concentrations of other indoor pollutants.

5. RESEARCH NEEDS

Substantial research efforts are needed in four interconnected areas: (1) study of radon sources and transport processes; (2) measure-

ment of the concentrations of radon and its daughters indoors and characterization of their behavior; (3) development and testing of control techniques; and (4) modeling of radon and its daughters in structures. These efforts need to be supported by development of measurement instrumentation and followed by an evaluation of control measures and building energy conservation measures. In addition, evaluative efforts require further work on the health effects of radon, which have not been discussed in this paper.

Programs to characterize building materials by radon emanation rate or radionuclide content should be more wide-scale and complete. Even more important, a program is needed to survey soil and groundwater in respect to radionuclide content, radon emanation, and radon transport. A rapid effort should be undertaken to determine the feasibility of geologic or geographic characterization of soil. As part of efforts to characterize materials, due attention should be given to the effect of moisture, pressure, and temperature. Community water supplies should also be surveyed.

Studies of indoor radon and daughter concentrations should be undertaken with two major purposes at this time: first, to learn the range and distribution of radon and/or its daughters in the building stock and, second, to understand the behavior of radon and its daughters in buildings. The first purpose requires surveys of a large number of buildings, covering a variety of building types and geographic areas. These surveys may be implemented by associating them with other wide-scale efforts, such as those for energy-conservation retrofits or for insurance purposes. These surveys may measure either radon concentra-

tions or PAEC, and the PAEC is the parameter of more direct health interest. However the radon concentration may be the preferable parameter for measurement in surveys, since an improved understanding of daughter behavior could then be used to infer PAECs in a way that is generalizable.

Such an interpretive basis must be developed through intensive measurements characterizing radon and daughter behavior indoors. Intensive work at only a few sites would serve as a basis not only for understanding measurement techniques but also for developing control techniques. In these studies, particular attention must be given to daughter particle interactions and removal processes. Results of these intensive investigations would be validated by less-detailed field measurements at a larger number of sites. Ultimately, these results would also serve as a partial basis for estimates of health effects.

Many such measurement programs will have to be supported by instrumentation development. More convenient portable instruments for field source measurements based on alpha scintillation techniques or on NaI gamma detectors could be developed. Further work on integrating devices for large-scale surveys of indoor concentrations is warranted, as is development of simple and quick daughter monitors with high sensitivity. For intensive investigation of daughter behavior at a few sites, more versatile special-purpose systems must be designed to automatically measure infiltration rate, radon, individual radon daughters, particle concentrations, and environmental conditions.

Substantial efforts to develop and study control techniques are required. The effects of techniques to clean the air (rather than

control the source) would have to be studied in the manner indicated above for detailed investigations of daughter behavior.

These measurement programs must be accompanied by corresponding modeling efforts. Models characterizing sources (on a geologic/geographic basis) and transport (by sites and building type) are needed. Although models for physical processes involving radon daughters have begun to be developed, much more work is needed, especially for understanding daughter-particle interactions and control techniques. Models of indoor air quality that appropriately utilize the source and daughter models could then be developed. Finally, the models of indoor air quality could be combined with models of the building stock in order to represent current radon and daughter concentrations and the effects of changes in building design and of potential control measures. Models of indoor air quality and the building stock will be necessary for assessing exposure to any indoor air pollutant and for evaluation of potential strategies for controlling indoor air quality.

6. CONCLUSIONS

The major factors affecting sources and levels of airborne radionuclides and radiation in buildings are known. However, for radon and its daughters, which are the major contributors to human radiation exposures indoors, concentrations vary greatly from one building to another and from one time to another. Although this variation is known to arise from differences in source magnitudes and in building type or operation, not enough is now known to characterize indoor concentrations generally, including the effect of specific radon sources or building features. Understanding present indoor concentrations, or the effect of potential

changes in building design, requires substantial further experimental and theoretical study of radon sorces, indoor concentrations and behavior, and control techniques.

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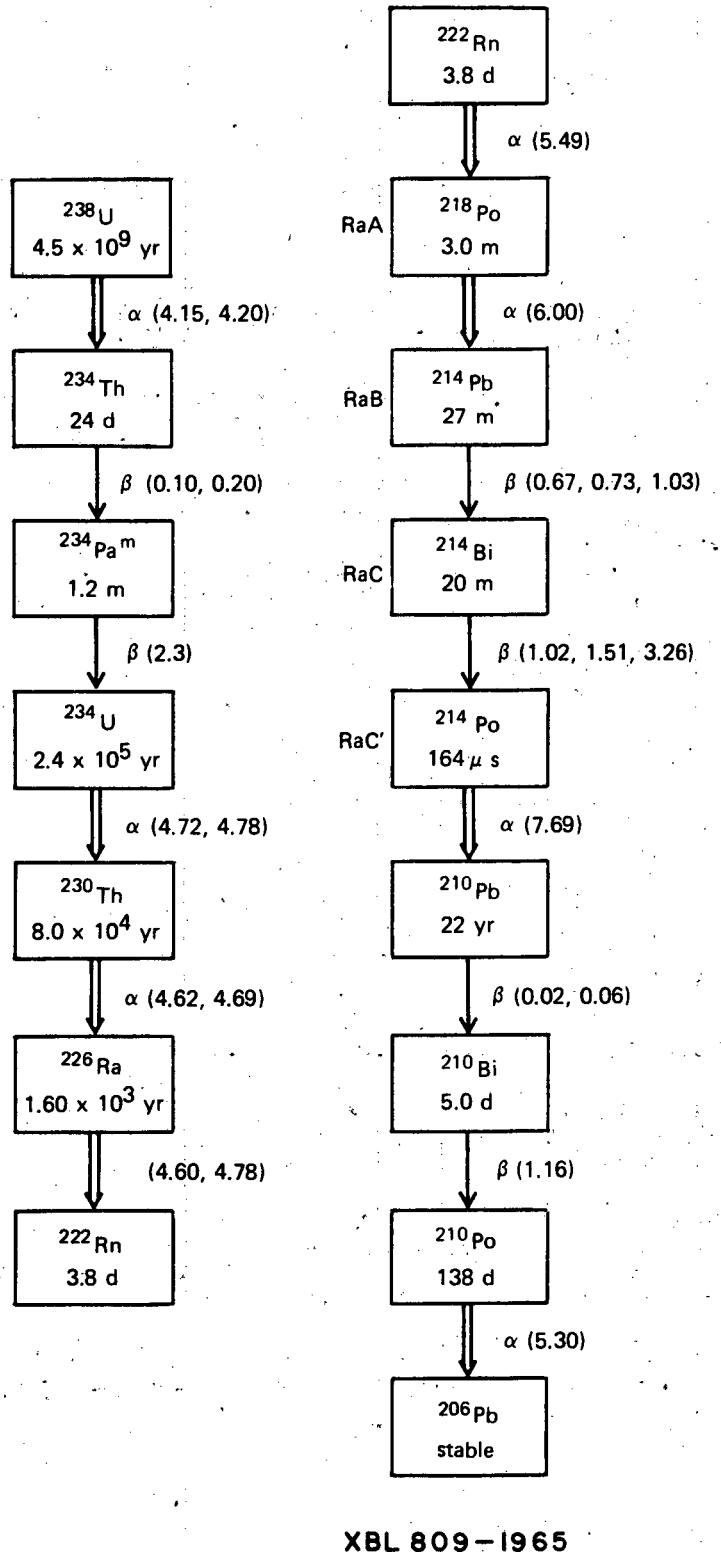

Figure 1. Uranium 238 decay sequence. The left portion shows the sequence from uranium 238 to radon 222, the right from radon 222 to lead 206; minor branches are not shown. Alpha decay is indicated by α, beta decay by β; decay energies are given in MeV (from Le78).

Table 1: Estimated Annual Tissue Absorbed Dose from Natural Sources in "Normal" Areas (adapted from UNSCEAR (Un77))

Source	Tissue absorbed dose rate (mrad/y)			
	Gonads	Lung	Bone lining cells	Red bone marrow
Cosmic radiation[a]	28	28	28	28
External terrestrial[b] radiation	32	32	32	32
Inhaled radon 222[c] and daughters	0.2	30	0.3	0.3
Other radionuclides in the body	18	23	25	31
Total	78	113	85	92
Fraction of absorbed dose delivered by alpha particles or neutrons (%)	1.2	31	8.5	2.1
Dose Equivalent[d] (mrem/y)	96	780[c]	220	130

[a]Outdoor dose rate; no structural shielding is presumed.

[b]Tissue dose rate approximately same outdoors and indoors; air dose is taken to be 18% higher indoors than outdoors, but different conversions to tissue dose compensate.

[c]Based on a representative indoor radon daughter concentration of 0.005 WL, in which case the total dose equivalent to the lung is dominated by alpha radiation from the daughters. This dose varies substantially among different types of tissue within the lung.

[d]Calculated assuming a quality factor of 20 (In77) for the high-LET dose (predominantly from alpha particles) and rounded to two significant figures.

Table 2. Average radionuclide content of U.S. building materials[a]

Reference	Material	^{238}U Series[b] (pCi/g)	^{232}Th Series[b] (pCi/g)	^{40}K (pCi/g)	Comments
L179	Concrete	0.29-1.32	0.28-1.58	6.6-9.8	Summarized measurements to select counting room materials
Ei78, Ka79	Concrete	1.4	1.5	21	Atlanta area
	Brick	1.8	1.8	17	Atlanta area
	Tile	1.9	1.1	8	Atlanta area
In81	Concrete	0.2-1.0	0.2-1.0	5-12	9 metropolitan areas
	Solar rock bed	1.5	1.4	25	New Mexico
Un77	Concrete	0.9-2.0	0.8-2.3	9-19	European concretes[c]

[a]Each entry is the average value for a sample group; a range is given for cases where several sample groups were examined.

[b]Because workers detected various members of the decay series, results in each column are directly comparable only if series equilibrium may be assumed. See indicated references for details.

[c]These values are taken, for comparison, from Un77, Table 8 (pg. 50), which gives a much more complete survey of European building materials than is available for the United States.

Table 3. Selected radon and radon daughter measurements in U.S. residences (residences are single family except where noted)

Location	Reference	^{222}Rn (pCi/l)[a]	Daughter PAEC (WL)[a]	Number of Residences	Type of Measurement	Comments
ORDINARY AREAS:						
Tennessee	Lo80		0.008(0.0008-0.03)	15	Grab	Shale area; mostly concrete construction
Boston	Ye72	0.07(0.005-0.2)	(up to 0.002)	7	Grab and ventilation	Single family; air exchange rate: 1-6 h^{-1}
		0.09(0.01-0.2)	(up to 0.002)	3	Grab and Ventilation	Multiple family; air exchange rate; 5-9 h^{-1}
NY/NJ	Geo78	0.8[b](0.3-3.1)	0.004[b](0.002-0.013)	21	Several integrated measurements over year	17 single family; 3 multiple family; 1 apartment bldg.
New York	F181	1.0(0.4-2.1)		11	Integrated winter[c]	Conventional houses
		6.4(2.0-26)		7	Integrated winter[c]	"Energy-Efficient" houses; ventilation rate not measured
Illinois	Ru79	(0.3-33)		22	Grab	Wood-frame construction, unpaved crawl spaces (windows closed)
San Francisco area	Be79	(0.04-0.8)		26	Grab and ventilation	Air change rate: 0.02-1.2 h^{-1} (windows closed)
U.S./Canada	Ho80	(0.6-22)		17	Grab and ventilation	Energy-efficient houses; air change rate: 0.04-1.0 h^{-1}(windows closed)
Maryland	Mo81	(0.1-27)	(0.001-0.12)	53	Grab and ventilation	Air change rate: 0.06-1.6 h^{-1}
SPECIAL AREAS:						
Grand Junction Colorado	Ba75		0.006[b]	29	Integrated year round	Controls for remedial action program (which has included houses in range 0.02-1 WL)
Florida	Gu78		0.004(0.0007-0.014)	26	Integrated year round	Controls on unmineralized soils
	Gu78		0.014 (up to 0.10)	133	Integrated year round	Houses on reclaimed phosphate lands
Montana: Butte	En80		0.02	56	Integrated year round	Intensive mining area
Anaconda	En80		0.013	16	Integrated year round	Intensive mining area

[a]Individual values are averages; values given in parentheses are ranges. All measurements are in living space; values in basements are typically higher.

[b]Geometric mean.

[c]Limited sampling indicates summer concentrations are approximately 20% of winter.

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