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by

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RADON AND ITS DECAY PRODUCTS IN INDOOR AIR - AN OVERVIEW

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ABSTRACT

This paper is an introduction to the question of radon 222 and its decay products in indoor air and a review of certain aspects of the question - for example, of results from surveys of radon concentrations in homes or of the potential importance of radon 220 and its decay products. Full scientific reviews of the major topics summarized here are to be found in the individual chapters of the book for which this serves as an introduction.*

Contents

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* W. W. Nazaroff and A. V. Nero (Editors), *Radon and Its Decay Products in Indoor Air*, in press (Wiley-Interscience).

I. INTRODUCTION

The radiation dose from inhaled decay products of radon 222 is the dominant component of natural radiation exposures of the general population. Monitoring in various countries yields average residential ^{222}Rn concentrations ranging from 10 to 100 Bq/m^3 (0.3-3 pCi/l: see appendix on quantities and units). For a country such as the United States, with an average of about 40 Bq/m^3 (1 pCi/l), the average lifetime risk of lung cancer caused by exposure to radon decay products is estimated to be about 0.3% (causing, then, on the order of 10,000 cases of lung cancer annually among the U.S. population of 235 million). As illustrated in Figure 1 - showing data from the United States - this average risk is more significant than that received on the average from all other natural radiation sources or from medical exposures. And the radon dose exceeds by a factor of ten to one hundred the average doses from nuclear power or weapons testing.

Moreover, in any country indoor levels a factor of 10 or more higher than the average sometimes occur. In fact, it is the common experience of the radon research community that ^{222}Rn concentrations in the range of 200 to 2000 Bq/m^3 (5 - 54 pCi/l) are found with startling frequency. And, although it might be thought that the estimated lung cancer risk associated even with an ordinary concentration such as 40 Bq/m^3 is very large compared with many environmental insults of concern, living for prolonged periods at concentrations above 200 Bq/m^3 leads to estimated individual lifetime risks exceeding 1%. The highest values found - more than 2000 Bq/m^3 - have risks even larger than those from cigarette smoking. However, unlike smokers, those living in unusually high radon concentrations are rarely aware of the large risks they may thereby be suffering.

The principal basis for present concern about exposures to radon's decay products is the experience with lung cancer incidence among underground miners. High fatality rates observed among miners as early as the sixteenth century were only later ascribed to lung cancer. And it was only after recognition in the 1950s that excessive lung cancer rates were occurring among uranium miners in the United States and elsewhere that examination of exposure history versus incidence indicated a relationship between added risk of lung cancer and exposures to ^{222}Rn decay products. Broadly speaking, results from a number of studies conducted among various miner groups, uranium and otherwise, have shown roughly consistent results. In the mean time, substantial efforts have been devoted to lowering exposure rates. Nonetheless, even current occupational limits - while keeping exposures much lower than before controls were implemented -

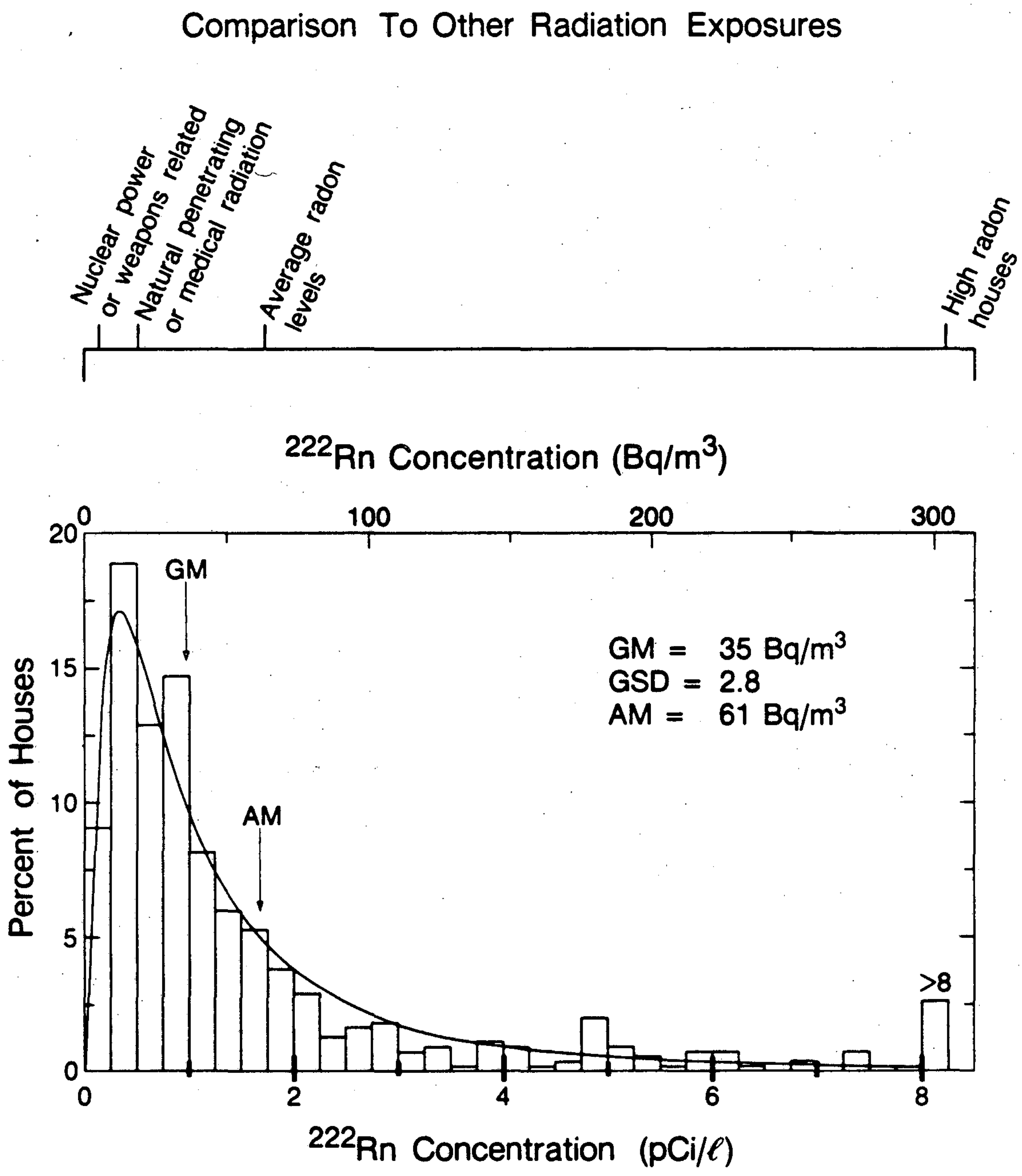

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Figure 1. Probability Distribution of ^{222}Rn in U.S. Homes

These data result from direct aggregation of 19 sets of data, totalling 552 homes, from the United States. The smooth curve is a lognormal function with the indicated parameters.[10] The upper scale indicates approximately the relative effective doses from radon and other sources of radiation exposure.

still entail a significant risk and, indeed, one that is larger than the risk associated with most occupational standards.

The experience gained in the uranium mines provided, not only information on health risks, but also the initial tools for understanding the occurrence, behavior, and control of radon and its decay products in the more general environment. The various isotopes of radon - ^{222}Rn the most important - are present to a greater or lesser degree in all environmental media - air, water, and soil - arising naturally from the radioactive decay of radium, whose isotopes are members of the decay series originating with uranium and thorium, primoridal constituents of the earth's crust.

The initial focus of environmental studies, however, was still exposures resulting from industrial processes, primarily the mining and milling of uranium, that increased the accessibility of radon to the outdoor atmosphere or to indoor environments. A principal example has been high ^{222}Rn concentrations inside homes and other buildings in the vicinity of Grand Junction, Colorado that were built on or with radium-rich tailings from uranium milling. Another case, in Canada, is that of the mining communities of Port Hope, Uranium City, Elliot Lake, and Bancroft, where remedial action programs were undertaken because of the possibility that indoor concentrations were unusually high because of local mining or milling activities. A comparable example is observation of higher-than-average indoor concentrations in homes built on lands in Florida that have been disturbed as the result of mining phosphate ores. These ores typically have elevated concentrations of the uranium series, and resulting waste products have high concentrations of radium. Although exposure limits were set by the United States and Canada for each of these cases, it is now clear that the concentrations found in these communities - while higher than average - are no higher than those occurring in other areas due merely to radon from the ground or, in some cases, from building components including unprocessed natural materials.

Indication of the potential significance of radon in the general building stock came with the realization in the 1970s of the health implications of very high concentrations in Swedish homes built using light-weight concretes incorporating alum shale as aggregate. This shale had extremely high radium content, causing high radon emanation rates from the finished concrete. This, together with low ventilation rates prompted by the interest in reducing energy use, resulted in high airborne concentrations in this segment of the housing stock. Ironically, it has subsequently been found in Sweden that the bulk of radon in the housing stock comes from the ground. Furthermore, monitoring in various countries has indicated that even average indoor concentrations are significant from the point of view of environmental risks. And, indeed, while changes in the ventilation rate can affect the indoor concentration significantly, the primary determinant of whether or not a particular indoor environment has high levels is, for many classes of buildings, the rate of radon entry.

Because of the apparent health implications, the early work on indoor concentrations has given rise to a broad range of research characterizing ^{222}Rn and its decay products indoors. This work has included significant monitoring programs in homes, investigation of the sources of indoor radon, study of the behavior of the decay products, and development of

Table 1

MAJOR COLLECTIONS CONTAINING INDOOR RADON ARTICLES[a,b]

Natural Radiation Environment III, proceedings of Symposium, Houston, Texas, April 1978 (Edited by T. Gesell and W.M. Lowder) - Technical Information Center (CONF-780422), Springfield, Virginia, 1980.

Radon and Radon Daughters in Urban Communities Associated with Uranium Mining and Processing, proceedings of three AECB Workshops in Ontario, Canada, 1978-1980 - Atomic Energy Control Board, Ottawa, Canada, 1978-1980.

Assessment of Radon and Daughter Exposure and Related Biological Effects, proceedings of Specialist Meeting, Rome, Italy, March 1980. (Edited by G.F. Clemente, A.V. Nero, F. Steinhausler, M.E. Wrenn) - RD Press, Salt Lake City, Utah, 1982.

Natural Radiation Environment, proceedings of Second Special Symposium on Natural Radiation Environment, Bombay, India, January 1981 (Edited by K.G. Vohra, U.C. Mishra, K.C. Pillai, and S. Sadasivan) - Wiley Eastern Limited, New Delhi, 1982.

Indoor Air Pollution, proceedings of the International Symposium on Indoor Air Pollution, Health and Energy Conservation, Amherst, Massachusetts, October 1981. (Edited by J. Spengler, C. Hollowell, D. Moschandreas, O. Fanger) - Special Issue of *Environment International*, *8*, (Nos 1-6), 1982.

Indoor Radon (Edited by A.V. Nero and W.M. Lowder) - Special Issue of *Health Physics*, **45** (No 2), August 1983.

Radon - Radon Progeny Measurements, proceedings of International Meeting, Montgomery, Alabama, August 1981 - U.S. Environmental Protection Agency (EPA 520/5-83/021), Washington, 1983.

Indoor Exposure to Natural Radiation and Associated Risk Assessment, proceedings of International Seminar, Anacapri, Italy, October 1983. (Edited by G.F. Clemente, H. Eriskat, M.C. O'Riordan, J. Sinnaeve) - *Radiation Protection Dosimetry*, **7**, (Nos 1-4), 1984.

Exposure to Enhanced Natural Radiation and Its Regulatory Implications, proceedings of Seminar, Maastricht, Netherlands, March 1985.(Edited by B. Bosnjakovic, P.H. van Dijkum, M.C. O'Riordan, and J. Sinnaeve)-Special Issue of *Science of the Total Environment*, **45**, October 1985.

Indoor Radon, proceedings of APCA International Specialty Conference, Philadelphia, Pennsylvania, February 1986 - Air Pollution Control Association, Pittsburgh (SP-54), 1986.

Indoor Air Quality, based on the Third International Conference on Indoor Air Quality and Climate, Stockholm, August 1984. (Edited by B. Berglund, U. Bergland, T. Lindvall, J. Spengler, and J. Sundell) - Special Issue of *Environment International*, **12**, (Nos. 1-4), 1986; more of the radon papers are in **Radon, Passive Smoking, Particulates and Housing Epidemiology**, Vol. 2 of *Indoor Air* (Edited by B. Berglund, T. Lindvall, and J. Sundell), Swedish Council for Building Research, Stockholm, 1984.

Radon and its Decay Products: Occurrence, Properties and Health Effects, proceedings of ACS Symposium, New York City, New York, April 1986 - American Chemical Society, Washington, to be published.

MAJOR REVIEWS

The Effects on Populations of Exposure to Low Levels of Ionizing Radiation (Committee on the Biological Effects of Ionizing Radiation[c]) - National Academy of Sciences, Washington, 1980.

Indoor Pollutants (Committee on Indoor Pollutants) - National Academy Press, Washington, 1981.

Ionizing Radiation: Sources and Biological Effects (United Nations Scientific Committee on the Effects of Atomic Radiation) - United Nations, New York, 1982.

Evaluation of Occupational and Environmental Exposures to Radon and Radon Daughters in the United States - Report No. 78, National Council on Radiation Protection and Measurements, Bethesda, MD., 1984.

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[a] For references on suggested radon standards, see A.V. Nero, "Elements of a Strategy for Control of Indoor Radon".

[b] Two major international meetings during 1987 will result in significant collections of indoor radon articles: The Fourth International Conference on Indoor Air Quality and Climate (Berlin, August 17-21, 1987) and the Natural Radiation Environment IV (tentatively planned for Lisbon in October). Both of these international conferences can be expected to continue on a periodic basis.

[c] The Committee is presently developing a report on alpha radiation.

techniques for controlling indoor concentrations. In addition, radiobiologists and epidemiologists have begun to apply dosimetric and dose-response data explicitly to the problem of environmental exposures. These international efforts have resulted in an extensive literature, including the major collections indicated in Table 1 and very many other papers scattered among a wide variety of journals and conference proceedings. The work for which this serves as an introduction is intended to be a substantive review of our growing understanding of radon and its decay products in indoor air.

Another result of the apparently large health implications of indoor radon has been substantial attention to the development of policies and strategies for preventing or eliminating excessive concentrations. Underlying this question is some specification, regulatory or advisory, of what constitutes "acceptable" versus "unacceptable" concentrations. Many who are newly initiated to the radon question are wont to refer to some particular agency's (or country's) "standard" for indoor radon. Most often the standard indicated is not a standard at all, but rather a guideline developed for some specific circumstance. (In rare cases, it actually is a regulatory standard, but narrowly drawn and not generally applicable to "indoor radon".) Similarly, a naive strategy for identifying and controlling excessive concentrations tends to rely on costly or inefficient survey or remedial techniques, as well as having no well-defined allocation of responsibility. Questions of standards and strategies are treated elsewhere. *

II. FUNDAMENTALS

A. Characteristics of Radon and Its Decay Products

A principal characteristic of radon that gives it more radiological significance than earlier members of the uranium (and thorium) decay chains is the fact that it is a noble gas. As such, once it is formed in radium-bearing material, a radon atom is relatively free to move, provided it first reaches the material's pore space (typically by recoil from the parent radium atom's emission of an alpha particle). Once in the pore space, macroscopic transport of radon is possible, either by molecular diffusion or by flow of the fluid in the pore space. Radon can therefore reach air or water to which humans have access, provided that transport is sufficiently rapid to be completed before the radon decays.

Radon 222 - formed in the ^{238}U decay chain (Figure 2) from decay of ^{226}Ra - is the most important radon isotope because it has the longest half life, 3.8 days. This is long enough that much of the ^{222}Rn formed either in building materials or in the ground within approximately a meter of building understructures can reach the indoor environment. Similarly, much of the ^{222}Rn formed within about a meter of the earth's surface reaches the outdoor atmosphere, although this has less radiological significance than that reaching indoor environments, which have relatively small volumes compared with the contributing source material. In some

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* A. V. Nero, "Elements of a Strategy for Control of Indoor Radon," Chapter 12 in Nazaroff and Nero.

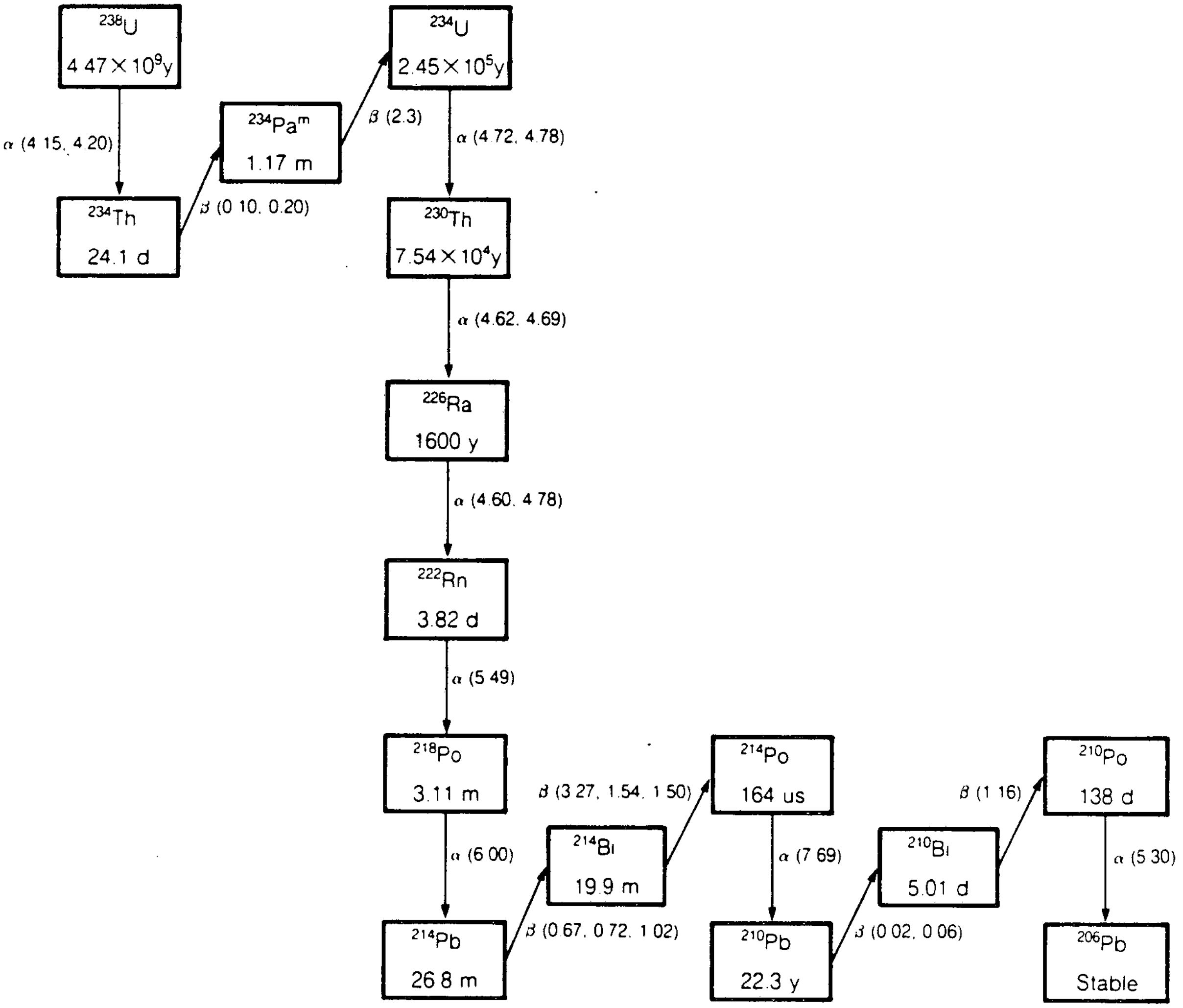

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Figure 2. Uranium 238 Decay Chain, including ^{222}Rn and its Decay Products

Radon 222, its parent ^{226}Ra, and its decay products are members of the ^{238}U decay series. Airborne concentrations of ^{218}Po, ^{214}Pb, and ^{214}Bi are of prime radiological interest due to their potential for retention in the lung, leading to subsequent irradiation by the alpha decays of ^{218}Po and ^{214}Po. See appendix for comparable ^{232}Th decay chain (given in Figure 9). (Figure courtesy of Lawrence Berkeley Laboratory.)

cases, whether for the indoor or outdoor environment, radon from much larger distances than a meter can be important if high-permeability transport routes (such as gravelly soil or fissures in the ground) are available.

In contrast, although about as much ^{220}Rn (sometimes called thoron) activity is formed (in this case from decay of ^{224}Ra, a member of the thorium 232 decay chain - see appendix D on ^{220}Rn), substantially less reaches air because its short half life (56 s) limits the distance it can travel before decay.* Finally, very little of another radon isotope, ^{219}Rn, is present in air because the ^{235}U decay chain, of which it is a member, has a natural abundance that is a factor of 100 lower and because of ^{219}Rn's short half life (4 s).

The second important characteristic of radon is that it decays to radionuclides that are both chemically active and relatively short-lived. As indicated in Figure 2, the four radionuclides following decay of ^{222}Rn have half lives of less than 30 minutes, so that - if collected in the lung on being inhaled - they are likely to decay to ^{210}Pb before removal by lung clearance mechanisms. (Similarly, ^{220}Rn begins a series of relatively short-lived isotopes; the most significant dose arises from the inhalation of ^{212}Pb, which has a half life of 10.6 h.)

The radiation released on decay of the short-lived decay products imparts the lung dose to which increased risk of lung cancer is attributed. The alpha radiation from the polonium isotopes contributes the radiologically significant dose, primarily because alpha particles deposit their energy within such a small thickness of tissue. As a result, the alpha energy is deposited in the relatively sensitive lung lining and also has a dense deposition pattern, which has much greater biological impact.

The concentration of (short-lived) decay products in air is ordinarily not given in terms of individual decay-product concentrations, but rather by a collective concentration that is normalized to the amount of alpha decay energy that will ultimately result from the mixture of decay products that is present. This quantity is the "equilibrium-equivalent" decay-product concentration (EEDC)⊕, the amount

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* Although indoor concentrations of the ^{220}Rn gas itself are ordinarily much less than those of ^{222}Rn, scattered data (cf. appendix on ^{220}Rn) suggest that, at least in buildings with average or below-average ^{222}Rn concentrations, the potential alpha energy concentration (PAEC) of ^{220}Rn decay products - while ordinarily less than that from ^{222}Rn decay products - can be a significant fraction of the total PAEC. Unfortunately, too little information is available to assess reliably the overall prevalence or importance of ^{220}Rn and its decay products, except to say that the PAEC has rarely, if ever, been found to approach the higher levels found for ^{222}Rn. As discussed later, however, results from measurements of indoor concentrations are consistent with a picture where ^{222}Rn transport is dominated by pressure driven flow, and ^{220}Rn transport by diffusion.

⊕ This quantity is often called the equilibrium-equivalent radon concentration, a practice that is avoided here because it is actually a measure of the decay-product concentration.

of each decay product necessary to collectively have the same potential alpha energy concentration (PAEC) that is actually present. (The decay-product concentration can also be given in terms of the PAEC itself. See appendix on quantities and units.) The ratio of EEDC to radon concentration is the equilibrium factor, equal to 1 if radon and all its decay products are in equilibrium (and therefore have the same radioactivity concentration), but in the range of 0.2-0.6 for most indoor atmospheres and somewhat higher outdoors.

For a given indoor radon concentration, the concentrations of the decay products can vary over a substantial range, since they are removed from the air, not only by radioactive decay, but also by ventilation and by reactions with the structure and its furnishings. An additional and important manifestation of their chemical activity is that the decay products can form small airborne agglomerates and can attach to previously existing particles. Such characteristics of the airborne decay products affect the rate at which they deposit on the walls and furnishings, the pattern and degree of deposition in the lung, and - ultimately - the magnitude and distribution of the associated radiation dose.

B. Factors Affecting Indoor Concentrations

The indoor concentration of radon and its decay products, or of any other airborne pollutant, depends on three factors: the entry or production rate from various sources, the ventilation rate, and the rates of chemical or physical transformation or removal. Because of its relatively long half life and lack of chemical activity, ^{222}Rn itself acts much like a stable pollutant whose indoor concentration is determined by only two factors, the entry rate and the ventilation rate. In contrast, the behavior of the decay products is much more complex, depending on the radon that is present, the ventilation rate, and the interplay among radioactive decay, chemical reactivity, particle concentrations, and the nature of the boundary layer between the indoor atmosphere and the surfaces that enclose it. Nonetheless, as a practical matter, the decay-product concentration is indicated approximately by the radon concentration, which is determined by source and ventilation characteristics. The influence of these factors on indoor concentrations is discussed here briefly. In addition, appendix B treats the question of ventilation rates *per se*.

Both excessive entry rates and decreased ventilation rates appear to be important causes of the high concentrations found in Swedish homes, which constituted a signal that scientists in other countries should investigate radon concentrations in their own buildings stocks. It is important to realize that extensive research on other aspects of indoor air quality was also beginning about the same time (see citations in Table 1), because of the discovery that several classes of pollution could occur indoors at higher levels than outdoors. For example, the byproducts of combustion - such as carbon monoxide, nitrogen oxides, and particulates - are the primary pollutants from the point of view of outdoor air quality. But concentrations are often much higher indoors, due to the presence of gas stoves, kerosene space heaters, and other appliances that are not vented effectively to the outdoors.[1,2] Similarly, although organic chemicals of various kinds are regulated in outdoor air and around toxic waste dumps, concentrations can be much higher indoors, again because of the presence of indoor sources.[3,4] In the United States, much of the research on indoor air quality began because of the fear that energy conservation measures might raise levels of such pollutants.

The basis of this concern may be seen by considering the steady-state concentration of a nonreactive gas entering an interior space from effectively internal sources (including the ground) at a fixed rate per unit volume, S_V. The concentration in this case is easily shown to be

$$I = (S_V + I_0 \lambda_V)/(\lambda_V + d), \qquad (1)$$

where λ_V is the ventilation rate, I_0 is the concentration in outdoor air, and d is the decay rate of the gas (if applicable). For typical ventilation rates, greater than 0.1 h^{-1}, the decay rate of ^{222}Rn (0.0076 h^{-1}) can be ignored in this equation, so that $I - I_0 = S_V/\lambda_V$. This simplifies even further, to $I = S_V/\lambda_V$, when - as is often the case - the outdoor concentration is much smaller than the indoor.

This expression does not encompass recently understood and important aspects of radon behavior, including time dependent and interactive phenomena, as discussed in later sections. Moreover, as is obvious, it cannot be used to describe the behavior of radon decay products, whose concentrations are determined by a number of interdependent reaction rates, as described qualitatively below and in more detail in succeeding chapters. Nonetheless, it is useful for specific purposes. An obvious example implication is that, for a fixed source strength, a decrease in the ventilation rate - e.g., to save energy - will result in a corresponding increase in the indoor concentration.

However, the relative importance of source strengths in determining indoor radon concentrations became clear in initial studies of concentrations and ventilation rates in U.S. homes. Earlier work had already indicated significant concentrations of ^{222}Rn and its decay products in U.S. homes;[5] and subsequent work had confirmed that, with supply of differing amounts of mechanical ventilation in a given house, the radon concentration varied as the inverse of the ventilation rate.[6] But results from simultaneous ^{222}Rn and ventilation-rate measurements in several groups of homes showed no apparent correlation between these two parameters.[7] As shown in Figure 3, the radon concentrations and ventilations rates for each sample group showed an approximately order-of-magnitude range; for the combined sample of about 100 homes, the concentration showed a significantly larger variability than the ventilation rate. If the ventilation rate λ_V and source strength S_V were independent in this sample of houses, and if the range of concentrations observed were caused primarily by differences in ventilation rate, then - because $I = S_V/\lambda_V$ takes a linear form, $\ln I = \ln S - \ln \lambda_V$, on a logarithmic plot - the data points of Figure 3 would cluster around a straight line with slope -1. No such correlation is apparent. The relative range and independence of these two quantities suggested that the source strength was the dominant determinant of the wide range of concentrations observed in U.S. housing.

Such indications have prompted substantial work in understanding the size and variability of radon entry rates, as discussed below. However, it is important to emphasize that other factors still play an important role: ventilation rates vary substantially within the housing stock,

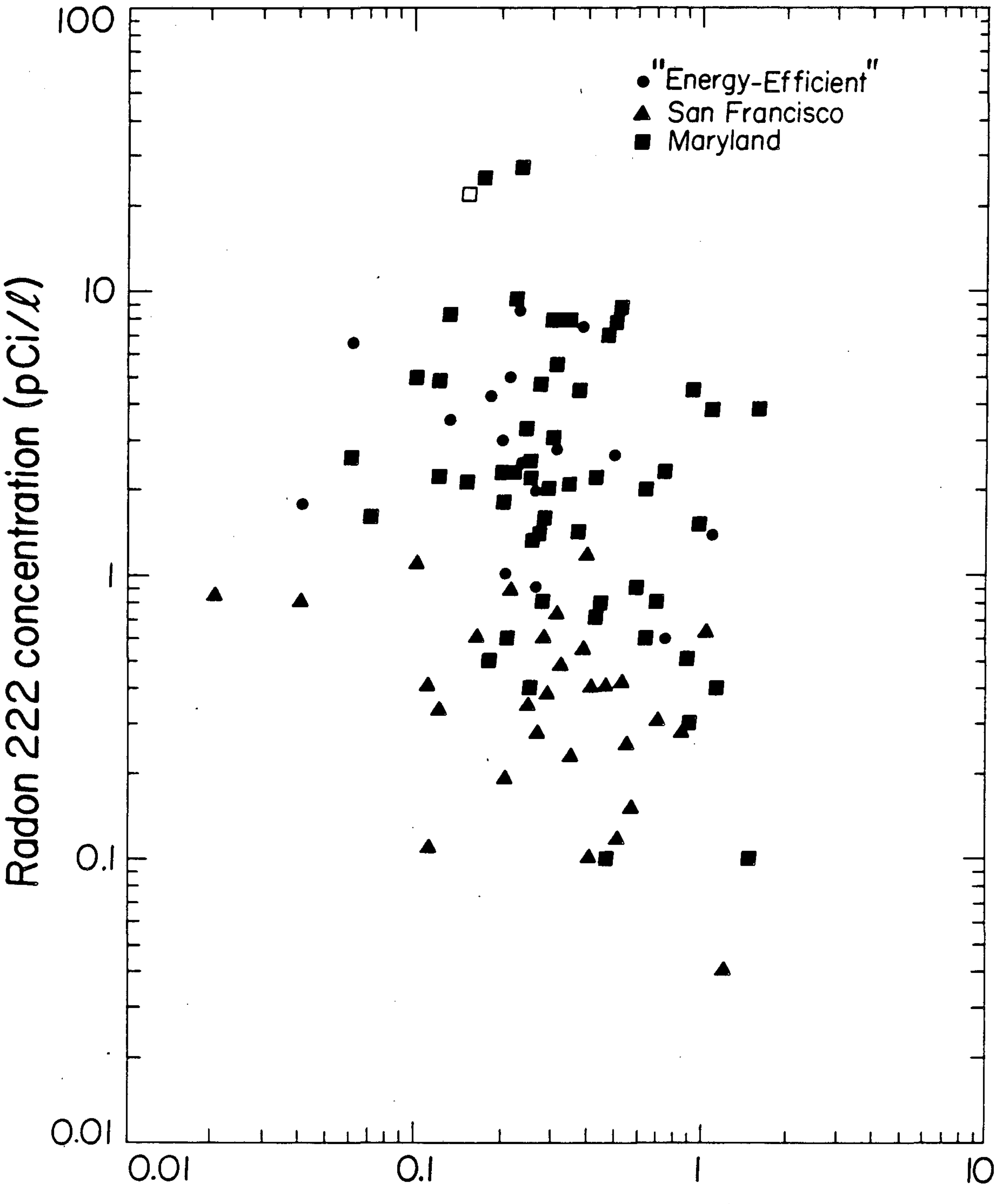

Figure 3. Radon 222 Concentrations and Air Change Rates Measured in 98 U.S. Residences

The results shown are from three survey groups: "energy-efficient" houses in the United States and (one) in Canada; conventional houses in the San Francisco area; and conventional houses in a community in rural Maryland (data from Ref. 7).

which is of course one of the major incentives for past and current efforts to increase the efficiency of energy use in buildings. (In the United States, an added incentive is that even the average ventilation rate - see Appendix B - is relatively high compared with rates in many countries.) And, even for a given radon concentration, the concentrations and physical state of the decay products - which account for the health effects of principal concern - can vary significantly. We turn now to a brief review of indoor ^{222}Rn concentrations, after which we will examine more closely the factors affecting indoor levels. Concentrations of ^{220}Rn and its decay products are treated in Appendix D.

III. INDOOR CONCENTRATIONS

Since identification of ^{222}Rn and its decay products as potentially important indoor pollutants, a large number of efforts have been undertaken actually measuring concentrations in homes. In a few countries, mostly European, these efforts have even included surveys in statistically-designed samples of the housing stock. Taken together, monitoring efforts to date provide a useful appreciation of residential radon concentrations in - not only Europe - but even the United States, where a wide variety of small or local survey efforts have been completed. Data are less complete for commercial and public buildings, although a variety of information can be brought to bear to suggest the approximate scale of radon concentrations in such environments.

This section briefly reviews the results of indoor radon surveys, primarily those performed in residences. No attempt is made to mention the many specific studies that have been performed. In particular, for residences - the focus of large numbers of efforts - only the few surveys or analyses that are representative of a large segment of the housing stock are discussed explicitly. Results from a significant number of individual surveys may be found in the major references given in Table 1 and in numerous individual journal articles and reports. Most of the results discussed here are summarized in Table 2.

It is also worth noting the tendency of the research community to measure radon concentrations in survey efforts, rather than decay-product concentrations. This tendency arises largely from the availability of reasonably reliable and very simple integrating etched-track radon monitors (see, for example, Ref. 8), a significant contrast to the state of decay-product monitoring. (An alternative technique being used in many shorter-term studies is based on collection of radon for a few days in charcoal cannisters; see, for example, Ref. 9.) Fortunately, given a reasonable understanding of the relationship between radon and its decay products and an awareness of the fact that the decay-product-to-radon ratio (typically in the range of 0.2 to 0.8) does not vary as widely as radon concentrations, measurement of the radon is a reasonable indicator of decay-product concentrations and is a very effective tool in survey efforts. This is analogous to the situation for many other pollutants: for example, although health effects associated with NO_2 exposures may have a substantial dependence on peak (rather than average) concentrations, an integrating sampler can be a very effective survey instrument, provided that associated studies examine relationships between average and

Table 2

Distribution of ^{222}Rn Concentrations in Various Countries

Country	No. of Homes Monitored	Type	Concentration (Bq/m^3) GM	AM	GSD or % of tail	Notes	Ref.
U.S.A	817	single-family	33	55	2.8	aggregated 22 data sets, adjusting to annual-avg.	10
	453		38	54	2.36	physics professors	11
Canada	9999		13		2.7	median values from 14 city surveys; mostly basement values; PAEC converted assuming equil. factor = 0.5.	14
Sweden	500	detached	69	122	2% > 800 Bq/m^3	built before 1975	15
		apartments	53	85	0.5% > 800 Bq/m^3		
Denmark	22	single-family		70		preliminary surveys; average of winter and summer means	18
		flats		82			
Finland	2000	small houses	64		2% > 800 Bq/m^3		17
Federal Republic of Germany	6000		40	49	1.8		20
Netherlands	1000		24		1.6	high levels show excess above lognormal	22
Belgium	79		41		1.7	preliminary national survey	79
France	765		44	76	2% > 200 Bq/m^3	incomplete natl. survey	80
United Kingdom	2000		15	25	2.6	living areas; bedrooms had GM of 11 Bq/m^3	21
Ireland	250		43		10% > 100 Bq/m^3	preliminary survey	81

peak concentrations under well-characterized conditions. As seen below, another incentive for emphasizing the radon concentrations per se is that this parameter appears to be a relatively direct indicator for use in strategies for identifying and controlling excessive concentrations and, as discussed elsewhere,* it may actually be a more consistent indicator of dose than the decay-product concentration.

A. Concentrations in North American Homes

Despite a broad range of U.S. efforts to characterize indoor radon, including a substantial number of studies that have included measurements in existing U.S. homes, there has been no direct broad-scale determination of the concentrations to which the U.S. population is exposed. The numerous U.S. studies have varied significantly in incentives, scientific objectives, selection of homes, and measurement procedures. The nature of the results from these studies therefore vary significantly, as do the conclusions that may be drawn from them. An obvious solution to this difficulty would be to carry out measurements in a valid statistical sampling of U.S. homes. From even an approximate evaluation of the data already in hand, it is clear that measurements in on the order of 1000 homes would determine the mean concentration very accurately and ascertain the fraction of homes at high concentrations (say, 10 times the mean) to a reasonable accuracy.

However, regardless of the potential importance of such an effort, the data already available are quite substantial, including tens of data sets from a wide variety of studies, with the precise number depending on the criteria for inclusion. It has been clear for some time that a systematic analysis of available data would have considerable value, and such an analysis has been performed on data available as of 1984-1985.[10] This analysis explicitly considered important differences between the studies and aggregated the various data sets, using both nonparametric and lognormal representations to yield a nominal radon distribution for single-family houses (the dominant type of residence) in the United States. Figure 1 shows the probability distribution obtained from direct aggregation of 19 of the data sets for which individual data were available. This particular aggregation, while not suitable for drawing conclusions about annual-average concentrations, indicates the substantial conformance of indoor radon concentrations to a lognormal representation, a result that has been observed in many individual studies.

The analysis examined a total of 38 data sets, corresponding to different areas (usually a state or urban area) of the United States. A significant number of these sets arose from monitoring efforts that were prompted by some prior knowledge of a potential for elevated concentrations. However, the main conclusions of the analysis were based on 22 data sets apparently lacking such prior indication. In either case (i.e., the 38 sets or the 22-set grouping), the geometric mean (GM) concentrations from the sets ranged from 11 to 210 Bq/m^3 (actually, 0.3 - 5.7 pCi/l, since all of these studies used the traditional units). Geometric standard deviations (GSDs) ranged from 1.3 to 4 among all 38 sets.

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* A. C. James, "Lung Dosimetry," Chapter 7 in Nazaroff and Nero.

The results of aggregating these sets in different manners are quite robust, with the main distinction being simply that aggregations utilizing the full 38 sets yielded a somewhat higher aggregate mean than those including only the 22 "unbiased" sets. The overall result, relying on the 22-set aggregations (and including a normalization of data taken only during heating seasons to obtain an estimate of year-long averages) is a distribution of annual-average ^{222}Rn concentrations that averages 55 Bq/m^3 (1.5 pCi/l) and that has a long tail with 1-3% of homes exceeding 300 Bq/m^3 (8 pCi/l). The analysis indicates that 7% (or about 4 million) of the U.S. single-family housing stock have concentrations greater than 150 Bq/m^3 (4 pCi/l). Alternatively, the distribution can be expressed as a lognormal function with a GM of 33 Bq/m^3 (0.9 pCi/l) and a GSD of 2.8. This result can only be associated with the portion of the housing stock consisting of single-family houses, since 99% of the data are drawn from such houses. These are, however, the dominant element of the U.S. housing stock, and the results indicate that on the order of a million homes in the United States have annual-average ^{222}Rn concentrations of 300 Bq/m^3 or more. (Another interesting observation from this analysis is that the geometric means of the "unbiased" 22 sets are themselves lognormally distributed, with a GSD of 2.0. This index demonstrates the substantial variability of mean indoor radon concentrations from one area to another.)

Currently, results from studies involving larger numbers of homes, often selected on a statistical basis or covering a larger portion of the United States, are beginning to become available. A notable example is a study that performed an integrated year-long measurement in the homes of approximately 450 physics faculty at about 100 colleges and universities across the United States.[11] The ^{222}Rn concentrations obtained average 54 Bq/m^3 (1.47 pCi/l) and fit a lognormal distribution admirably well, with a GM and GSD of 38 Bq/m^3 and 2.36, respectively. This implies that 0.8% of homes have concentrations greater than 300 Bq/m^3, at the low end of the range estimated from the analysis discussed above. On the other hand, the fraction above 150 Bq/m^3, 6%, is very close to the earlier result. Although a study of homes of physics faculty cannot be construed to be a representative sampling of U.S. homes (a problem suffered also by the aggregate analysis just discussed), this study has the advantage that the homes are widely distributed across the United States and that the measurement protocol determined directly a year-long average concentration. The averages from the 100 or so institutions are themselves lognormally-distributed, with a GM of 46 Bq/m^3 and a GSD of 1.75. Interestingly, the highest average from both this study and the aggregate analysis above occurred in the state of North Dakota.

This suggests the need to recognize that the broad distributions discussed here do not of themselves reveal the much higher distributions that may occur in specific areas. For example, both studies just referred to indicate that areas of North Dakota have average concentrations much higher than the U.S. mean, and a correspondingly higher portion of houses exceeding any particular concentration. More recently, two other such areas have been identified. One is in the Spokane River Valley of Washington and Idaho, where the average winter concentration in 46 homes was found to be 500 Bq/m^3 (13.3 pCi/l), with 20 of the houses exceeding 300 Bq/m^3.[12] At about the same time, astounding concentrations were found in eastern Pennsylvania (a state that was already known to have very high

high concentrations.[16]

Preliminary results from measurements in more than 2000 houses in Finland yields a GM of 64 Bq/m^3, with about 2% exceeding 800 Bq/m^3, similar to the Swedish single-family results;[17] a strong geographical dependence was observed. Results from a small sampling in Denmark show somewhat lower concentrations in that country, e.g., average (not GM) concentrations in single-family dwellings of 88 Bq/m^3 in winter and 24 Bq/m^3 in summer;[18] however, some of the apparent difference may arise from differences in measurement period.

Results from a survey of about 6000 homes in the Federal Republic of Germany using etched-track ^{222}Rn detectors have recently become available, yielding a GM of 40 Bq/m^3 and GSD of 1.8.[19] In spite of the fact that a larger number appeared at high concentrations than indicated by these lognormal parameters, only 10 (0.2%) of the homes sampled had concentrations exceeding 500 Bq/m^3. Other results from the FRG indicate an equilibrium factor of 0.3 indoors and 0.4 outdoors.[20]

A survey mounted in the United Kingdom has measured ^{222}Rn concentrations in about 2000 homes, yielding a GM of 15 Bq/m^3 and GSD of 2.6 for living areas (11 Bq/m^3 and 2.5 for bedrooms). The ratio of summer to winter concentrations was found to be 0.51 in living areas and 0.53 in bedrooms. Concentrations substantially higher than these overall values have been found in specific areas, such as Cornwall.[21]

Measurement of ^{222}Rn concentrations in about 1000 dwellings in the Netherlands gave a distribution well represented (except for an excess of high concentrations) by a lognormal function with GM 24 Bq/m^3 and GSD 1.6.[22]

In Europe, it is also important to realize that many of the countries mentioned have identified areas or portions of the housing stock with exceptionally high concentrations. As noted earlier, it was housing with concretes using alum-shale as aggregate that drew initial attention to indoor radon as a problem, but more recently radon from the ground has been identified as the primary source in Swedish housing, and this varies substantially from one area to another. In Finland, a study on the south coast found averages in the 31 locales surveyed to range from 95 to 1200 Bq/m^3, with the total sample averaging 370 Bq/m^3 and 12% of houses exceeding 800 Bq/m^3.[23] As a final example, the average from Cornwall was 390 Bq/m^3, fifteen times the national average for the United Kingdom.[21]

More limited information is available from a number of other countries. Except for special local areas, none of the results exceed the distribution found for the Scandinavian countries. And some approach the considerably lower levels characteristic of the outdoors, with average concentrations in the vicinity of 10 Bq/m^3.

C. Other Buildings

Very few data are available on radon concentrations in buildings other than residences. However, the same range of radon sources contributing to residential radon also contribute - to a greater or lesser levels in some locales).[13] Although no dependable frequency distribution can be extracted from the results from monitoring about 4000 homes in this area, the average appears to be similar to the Spokane results, but with a much larger number of houses involved and with indoor radon concentrations as high as 100,000 Bq/m^3. An exceptional concentration of high levels has also been found in the town of Clinton, New Jersey, where about half of the 100 or so houses monitored in one neighborhood had winter concentrations exceeding 7500 Bq/m^3.

A significant number of the earlier studies analyzed in Ref. 10 measured the concentration of decay products instead of radon. One of these in particular[5] measured both concentrations averaged over week-long periods, thereby determining directly an effective equilibrium ratio, which was found to range from 0.31 to 0.82 among the houses examined. As will be evident from the measurements from other countries, discussed below, there is significant evidence that equilibrium ratios tend to average about 0.4 among samples of housing, with significant variability from one time to another, from one house to another, and perhaps some differences from one region or country to another.

One of the earliest large studies was conducted in Canada, where an approximate idea of indoor concentrations has been obtained by performing grab-sample measurements in a large number of homes (9,999).[14] Rather than being a representative sampling, homes were selected from 14 of Canada's largest cities, therefore directly representing a large portion of the housing stock. The geometric mean EEDC varied from 3 to 13 Bq/m^3 among these cities, the median being 7 Bq/m^3. The median GSD was about 2.7. These results cannot, however, be thought of as representing the living space of these homes, since the preferred sampling site was chosen to be the basement. (The ^{222}Rn concentrations were also measured, but are suspect in view of the very large GSD obtained, a median of 4.0 among the 14 cities, and the fact that measurements were performed in basements.)

B. Residential Concentrations in Europe

Significant efforts to determine indoor radon concentrations have been conducted in numerous European countries. However, in only a few have these efforts been designed to provide representative information on concentrations in the housing stock as a whole. The results from studies designed more or less to meet this objective differ to a significant, but understandable, degree from one country to another.

Sweden, a principal in inciting the wide interest in radon, has more recently completed a country-wide survey, utilizing two-week passive ^{222}Rn measurements in some 500 homes built before 1975.[15] The results averaged 122 Bq/m^3 (GM 69 Bq/m^3) for 315 detached houses and 85 Bq/m^3 (GM 53 Bq/m^3) for 191 apartments. The distributions are approximately lognormal, with 10% of the results over 266 Bq/m^3 (detached) or 187 Bq/m^3 (apartments). Overall, concentrations are higher than for countries outside Scandinavia, with about 2% of detached homes and 0.5% of apartments estimated to have ^{222}Rn concentrations exceeding 800 Bq/m^3 - comparable to their remedial action level of 400 Bq/m^3 EEDC, assuming a typical equilibrium factor of 0.5. A supplementary study of detached houses built since 1975 yielded a GM similar to that for the larger study but with a smaller fraction at degree - to levels in other buildings. Hence, it is to be expected that the minimum concentration observed in such buildings is the outdoor level - averaging about 10 Bq/m^3 of ^{222}Rn, as indicated in appendix C - but that higher levels, even into the range of those in residences, should be the norm. To the extent that structures such as hospitals or office buildings have a larger number of stories and higher ventilation rates than residences, we would expect lower concentrations.

Only a small number of measurements in other buildings - whether offices, stores, hospitals, or schools - have actually been performed, and virtually no statistically meaningful data have been developed. It is, nonetheless, worth mentioning two substantial studies from the United States as examples. One of these was conducted in the Pittsburgh, Pennsylvania area, using grab-sample measurements taken during occupancy (an important issue if ventilation systems are turned off during unoccupied hours) in schools, stores, and other public and commercial buildings.[24] Average concentrations for various groups of buildings, many including on the order of a hundred measurements, were in the vicinity of 15-20 Bq/m^3, slightly higher than the outdoor concentrations measured (about 10 Bq/m^3), and much less than the annual-average concentration found in a local survey of residential levels, about 96 Bq/m^3. A survey of 38 office buildings in the Pacific Northwestern United States found concentrations to average 11, 26, and 44 Bq/m^3 for the three cities included.[25] These results, obtained with integrated samplers, are somewhat lower than comparable residential concentrations in the respective areas.

D. Time Dependence of Indoor Concentrations

Numerous measurements of radon in dwellings have indicated the substantial variation with time of ^{222}Rn and its decay products. This variability occurs with time of day, weather conditions, or season. Without ascribing any cause to this variability - which is certainly related to environmental or operational parameters affecting radon entry or removal (or behavior of the decay products) - it is useful at this point to give one or two examples.

One of the earlier examples was obtained using continuous ^{222}Rn monitors in a New Jersey house, measuring basement and upper floor concentrations over a period of weeks.[26] A significant diurnal dependence was observed, consistent with later measurements, which however sometimes showed a stronger difference between maxima and minima.

More recent work has examined the dependence of concentrations on various factors, directly measuring not only the concentration, but also source parameters and environmental factors affecting radon entry and ventilation rates.[27] An example of such real-time measurements is shown in Figure 4.

Detailed examination of such correlations is important for understanding radon entry and removal, as discussed later in this paper and

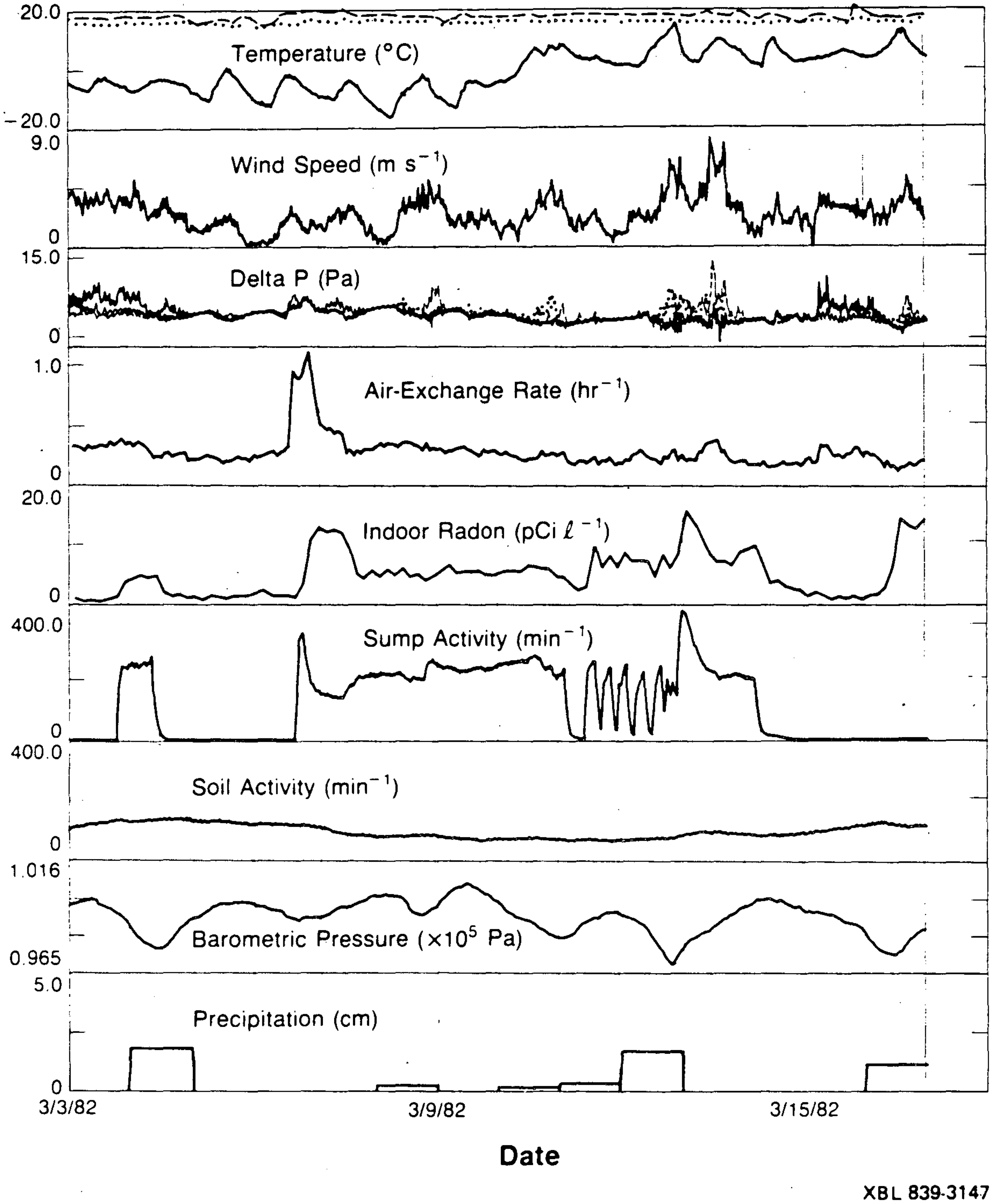

Figure 4. Variability of ^{222}Rn Concentration, Ventilation Rate, and Environmental Parameters in a House with a Basement

These data were accumulated over a two-week period during a several-month experiment examining the dependence of radon entry on environmental factors.[27]

elsewhere.* An appreciation of variability is also a key element in proper interpretation of monitoring results and effective selection of control measures.⊕

IV. SOURCES AND TRANSPORT

Radon arises from trace concentrations of radium in the earth's crust, and indoor concentrations depend on access of this radon to building interiors. Radon can enter directly from soil or rock, via utilities such as water (and, in principle, natural gas) that carry radon, or from crustal materials that are incorporated into the building structure in the form of concrete, rock, and brick. The relative importance of these pathways depends on the circumstances, but it has become clear that the first - direct ingress from the soil - ordinarily dominates the higher indoor concentrations that have been observed in homes.

Indications of this arose in early investigations of radon in U.S. houses, when it was found that measurements of radon emanating from structural materials could not account for observed indoor concentrations, based on estimates of the air exchange rate.[5] Moreover, practical experience in reducing concentrations in the Canadian mining communities made it clear that the major entry route was through the house understructure, at least in the houses in which remedial measuers were attempted.

A clearer picture emerged from the distribution of entry rates inferred from direct measurements of radon concentration and ventilation rate (such as the data shown in Figure 3). Figure 5 shows such entry-rate distributions from various countries, as well as indicating the potential contribution of various sources. Although building materials were first suspected as the major source in the U.S., based on the experience in Europe (e.g., Ref. 28), the initial U.S. results strongly suggested that the soil must be the major source.[7] Understanding how the rate of radon entry could be approximately equal to the unimpeded flux from the ground (i.e., in the absence of the house) has been a major focus of research on radon entry, in both North America and Europe.

This section discusses briefly the current state of knowledge of radon sources and transport. Far more comprehensive treatments are given in Chapters 2, 3, and 4 of Nazaroff and Nero.

A. Soil and Building Materials

Understanding the radon mass balance for a building requires specific consideration of the various sources. As indicated in Figure 5, a median (or GM) entry rate for U.S. single-family houses appears to be in the vicinity of 20 Bq $m^{-3}h^{-1}$ (0.5 pCi $l^{-1}h^{-1}$). Based on emanation rate measurements from U.S. concretes,[29] one might expect emissions from this source to account for a median of about 2 to 3 Bq $m^{-3}h^{-1}$, far below the

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* W. W. Nazaroff, B. A. Moed, and R. G. Sextro, "Soil," Chapter 2 in Nazaroff and Nero.

⊕ A. V. Nero, "Elements of a Strategy for Control of Indoor Radon," Chapter 12 in Nazaroff and Nero.

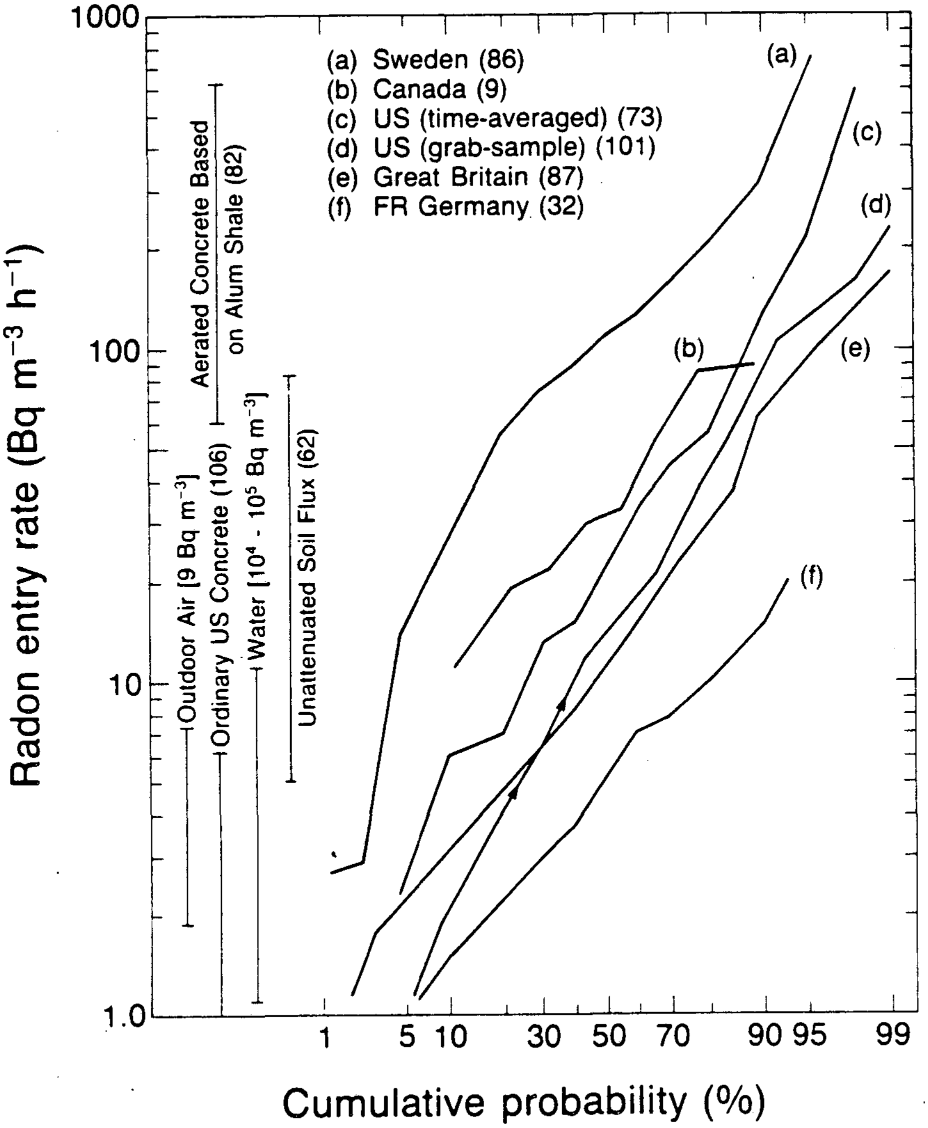

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Figure 5. Cumulative Frequency Distributions of Radon Entry Rates in Dwellings

The entry rate is determined from the product of simultantously measured radon concentration and ventilation rate. The number of residences in each sample is indicated in parentheses. The references from which these results are taken are: (a) 28 (b) 71 (c) 6, 27, 35, 72-74 (d) 7 (e) 75 and (f) 76. The bars at the left indicate the range of contributions expected from a variety of sources, with assumptions indicated in brackets. For each source, we have assumed a single-story house of wood-frame construction with a 0.2-m-thick concrete-slab floor. The floor area and ceiling height are assumed to be 100 m^2 and 2.4 m, respectively; water usage is assumed to be 1.2 m^3 per day, with a use-weighted transfer efficiency for radon to air of 0.55; and the ventilation rate is assumed to be in the range 0.2-0.8 h^{-1}. References for estimates of source contributions are: outdoor air (62), U.S. concrete (29), alum-shale concrete (66); water (31); and soil flux (30). (Figure taken from Ref. 77).

rate observed. On the other hand, the potential contribution from unattenuated soil flux, a median of about 25 Bq $m^{-3}h^{-1}$ (based on Ref. 30), corresponds much more closely with the indoor observations. However, houses have understructures that might be expected to impede substantially the ingress of radon, at least by diffusion, the main mechanism for causing the observed fluxes from uncovered soil. As discussed in detail in Nazaroff *et al.** and in Ref. 31, although transport via diffusion accounts well for observed fluxes from building materials and exposed soil, and could account for small fluxes from the soil through some understructure materials (such as concrete), diffusion cannot account for the total entry rates observed in single-family houses. Another mechanism must account for the efficiency with which radon from soil enters such homes. It appears that this mechanism is bulk flow of soil gas driven by small pressure differences between the lower part of the house interior and the outdoors.

As discussed in Appendix B on ventilation rates, pressure differences of only a few pascal - i.e., on the order of only a ten thousandth of an atmosphere - arise from winds and indoor-outdoor temperature differences and are the cause of ventilation in homes during seasons when the windows are closed. These same pressure differences can, in principle, drive the small flows of soil gas needed to account for the observed rate of radon entry into homes: soil gas contains enough radon that, on the average, only 0.1% of infiltrating air would have to be drawn from the soil to account for observed indoor concentrations.[31]

Recent work has begun to characterize directly and systematically the potential for pressure differences to cause entry of radon via soil gas, probably through imperfections and penetrations in the house understructure that permit passage of the relatively small amount of soil gas required. A study of radon entry in a single family house with a basement(see Figure 4) analyzed the entry rate versus the ventilation rate, measured over a period of months, and concluded that entry could usefully be represented by a sum of two components: one - the smaller - independent of ventilation rate, much as diffusion would be, and a larger term proportional to ventilation rate, as pressure-driven flow might be.[27] Moreover, the observed pressure and soil parameters appeared consistent with the soil-gas flow rate that was implied by the measured concentrations and ventilation rates. In addition, theoretical simulations of transport (e.g., Ref. 32) are helping to form a fundamental picture. Finally, recent experiments have directly observed, in houses with basements, the underground depressurization implied by this picture, and have monitored underground soil-gas movement by injecting and monitoring tracers.[33] A different kind of experiment - practical experience with remedial measures in such areas as Eastern Pennsylvania[34] and the Spokane River Valley[12] using techniques of the kind discussed in Scott[θ] is confirming and providing new information on the flow of radon-bearing air through house understructures. It is interesting to note that these results may also have significant implications for entry of other pol-

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* W. W. Nazaroff, B. A. Moed, and R. G. Sextro, "Soil," Chapter 2 in Nazaroff and Nero.

θ A. G. Scott, "Preventing Radon Entry," Chapter 10 in Nazaroff and Nero.

lutants from the soil.

The studies of houses with basements have given results that may also apply in large part to slab-on-grade structures, where the pressure difference generated can still draw soil gas through any penetrations in or around the slab. However, few direct measurements in such structures have been performed. The other understructure type of substantial importance is the crawl space, which to some extent isolates the interior from the soil - at least in respect to pressure-driven flow between the two. Limited measurements of the transport efficiency of radon through crawl spaces yield the result that a substantial portion of the radon leaving uncovered soil manages to enter the interior, even if vents are open to permit natural ventilation of the space.[35]

In retrospect, this is not entirely surprising, since the stack effect will tend to draw infiltrating air into the home from the crawl space, which can retain radon from the soil in conditions where winds are not sufficient to flush it to the outdoors via the vents. Furthermore, for structures where the vents are sealed shut, e.g., to save energy, it is conceivable that the crawl space still provides sufficient connection between the house interior and the soil that pressure-driven flow can enhance the flux from the soil above levels associated merely with diffusion; the work reported in Ref. 35 may have observed this effect. Another observation from this study in that energy conservation efforts that focus on tightening the floor above a crawl space can significantly reduce infiltration rates, while reducing radon entry a corresponding amount, as a result of which indoor concentrations are little affected.

Thus sufficient mechanisms exist to account for the substantial amount of radon that appears to enter single-family homes from the soil, apparently without great regard to substructure type. However, this does not imply that other sources of radon are unimportant. As discussed elsewhere,* it is clear that materials utilized in a building structure can contribute substantial indoor concentrations, although this is not usually the case (even for natural stone that has higher-than-average radium content). Moreover, in buildings that are relatively isolated from the ground, such as multi-story apartment buildings, indoor concentrations are expected to be lower than average - as if often the case in central European dwellings - and to arise primarily from the building materials and from radon in outdoor air.

B. Water

Probably more important than buildings materials, as a source of radon in certain parts of the housing stock, is domestic water drawn from underground sources. Surface waters have radon concentrations too small to affect indoor levels when used domestically, but ground water is in a good position to accumulate radon generated within the earth's crust. As a result, very high radon concentrations can sometimes be found in associated water supplies. For example, a survey of ^{222}Rn concentrations in well-water in Maine found a range of 7×10^2 to 7×10^6 Bq/m^3 (20 to

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* E. Stranden, "Building Materials," Chapter 3 in Nazaroff and Nero.

180,000 pCi/l), while wells in granitic areas were found to average 8 x 10^5 Bq/m^3;[36] and the average concentration in water from drilled wells in Finland has been found to be 6.3 x 10^5 Bq/m^3.[36a] With normal water use, the radon entering indoor air from water with such high concentrations can be expected to be significant.

Examinations of the overall potential contribution of water-borne radon to indoor concentrations have tended to be no more sophisticated than to consider typical water use rates, house volumes, and ventilation rates, yielding a ratio of radon in air to radon in the water supply of about 10^{-4}. However, it is possible to make a more realistic estimate of the distribution of the air-to-water ratio, using more detailed distributional information (rather than just averages) on water use rates, efficiency of radon release from domestic water used in various ways, house volumes, and ventilation rates. Such an analysis of U.S. data for single-family houses yields a ratio with a lognormal distribution having a GM of 0.65 x 10^{-4} and a GSD of 2.88, as discussed elsewhere.* Taken together with recently developed data on ^{222}Rn in public water supplies,[37] such a distribution permits quantitative assessment of the contribution of public water supplies to indoor radon concentrations. The result of such an assessment is that such supplies contribute an average of approximately 1 Bq/m^3 (0.03 pCi/l) in the 30% of U.S. homes served by ground water, only about 3% of the average indoor concentration in U.S. single-family homes.

However, the very high water-borne concentrations that are sometimes found - particularly from private wells - must be expected to contribute much larger airborne concentrations in the homes affected. Using the few data that are available for the approximately 18% of the U.S. population using private wells, the indoor contribution from water for this segment of the housing stock can be estimated to average about 20 Bq/m^3. About 10% of the houses served by these wells (totaling about 2% of the U.S. housing stock) are estimated to have indoor concentrations from water of 40 Bq/m^3 or more. Although these estimates for radon from private wells cannot be regarded to be reliable, they suggest that the portion of the population using private wells may be experiencing significantly higher radon exposures than average, particularly in areas with high radon activity in water.

C. Summary of Source Contributions

We can thus point to the major sources of radon present in indoor environments. For single-family houses and other structures of one or two stories, the ground constitutes the principal source, but with a noticeable portion entering with outdoor air, i.e., on the order of 10 Bq/m^3. However, there are circumstances in which the building materials or domestic water supplies are important. In contrast, in high-rise buildings, including apartments, the ground is of lesser importance, and the amount of radium-bearing building materials per unit volume is larger, so that outdoor air and the building materials are typically the dominant contributors, although the ground and water supplies can be important in some

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* W. W. Nazaroff *et al.*, "Water," Chapter 4 in Nazaroff and Nero.

cases. Other potential sources, such as natural gas ordinarily contribute negligible amounts by comparison.[38,39]

For a specific set of assumptions, characteristic of U.S. buildings, Table 3 indicates the approximate average contribution of various sources to the indoor concentration for single-family buildings and (to the extent data are available) for apartments. For comparison, the observed concentrations are also given. Note that, considering the uncertainties involved, the presumed contributions are consistent with the observed levels, i.e., an average of 55 Bq/m^3 in single-family houses and a substantially smaller concentration in high-rise apartments.

V. BEHAVIOR OF THE DECAY PRODUCTS

Even for a given amount of ^{222}Rn, the concentrations of its decay products and their physical state can vary considerably. What particularly distinguishes the decay products from ^{222}Rn itself is their chemical activity: the decay products can attach to airborne particles, to indoor (macroscopic) surfaces, and to the human tracheobronchial tract, where they can deposit either directly or after attachment to airborne particles. In addition, the detailed behavior and health significance of the decay products is greatly influenced by their half lives and decay modes, indicated in Figure 2. The alpha decays imparting the radiation dose of greatest significance are those of ^{218}Po and ^{214}Po, for the ^{222}Rn series. (The comparable isotope for the ^{220}Rn series is ^{212}Po, as indicated in appendix D.) The overall concentration of decay products, given by the potential alpha energy concentration (PAEC) or the equilibrium-equivalent decay-product concentration (EEDC), depends on the concentrations of the first three decay products (^{218}Po, ^{214}Pb, and ^{214}Bi) for the 222 series (and on ^{212}Pb and - to a minor extent ^{212}Bi - for the 220 series) and on the amount of polonium alpha energy that each will yield (see appendix A on quantities and units).

Understanding the behavior of the decay products is needed, not just to satisfy a general scientific interest in the relationship between indoor radon concentrations and decay-product concentrations, but specifically to provide a quantitative basis for characterizing the radiological implications of decay-product exposures and for indicating the potential utility of control measures aimed at modifying concentrations of the decay products. Although the discussion of this section focuses primarily on behavior of ^{222}Rn decay products, the same considerations apply in the main to the ^{220}Rn series - at a few points in the discussion, critical differences are indicated

A. General Considerations

In an atmosphere where the concentration of radon is stable and where each decay product, once formed, is removed only by radioactive decay, radon and its decay products would be in a state of equilibrium, all having the same (radioactivity) concentration. In an indoor atmosphere where air is exchanged at some fixed ventilation rate, not only is the radon concentration less than it would otherwise be, but the concentrations of the decay products are reduced relative to that of radon because

Table 3. Approximate Contributions of Various Sources to Observed Average Radon Concentrations[a]

	Single-Family Houses		High Rise Apartments	
	Bq/m^3	(pCi/l)	Bq/m^3	(pCi/l)
Soil potential (based on flux measurements)	55	(1.5)	>0	(>0)
Water (public supplies)[b]	0.4	(0.01)	0.4	(0.01)
Building materials[c]	2	(0.05)	4	(0.1)
Outdoor air	10	(0.25)	10	(0.25)
Observed indoor concentrations	55	(1.5)	12?	(0.3?)

[a] In each case, the arithmetic mean is shown, based on entry rates (or, in the case of outdoor air, concentrations) discussed in the text and on an air exchange rate of 0.5 h^{-1}.

[b] Indicated water contribution applies to 80% of U.S. population served by public supplies. Contribution may average 20 Bq/m^3 in homes using private wells, with even high contribution in high-activity areas.

[c] The contribution of building materials in single-family houses corresponds to a slightly smaller geometric mean entry rate than that estimated in the text because not all houses are slab-on-grade or one-story. A higher contribution to apartment air is suggested on the presumption that, on the average, high-rise apartments have a larger amount of radium-bearing material per unit volume than do single-family homes.

of removal by ventilation.

However, the behavior of the decay products, as suggested above, is further complicated by their chemical activity: the fact that the decay products can attach to particles or surfaces, and that these attachment rates can vary with conditions, makes general characterization of the state of the decay products - and of its dependence on ventilation rate, particle concentrations, and other factors - exceedingly complex. However, since we are dealing with only a few species, whose rate of production from early members of the decay chain is determined solely by known half lives, it is still possible to specify a relatively straightforward framework for considering the behavior of the decay products.

Figure 6 illustrates, for an unspecified decay product (or "progeny"), various mechanisms for changing the state or presence of the decay product, other than radioactive decay itself. Because the deposition rates for the decay products depend strongly on whether or not they are attached to particles - and even on the particle characteristics - airborne particles play a crucial role in determining the concentrations that are present in air, and potentially on the radiation dose that results from a given concentration. Given the parameters that are indicated in Figure 6, one can write down a system of conservation-of-mass equations, following Ref. 40, that determine the concentrations based on given rate constants or - conversely - that can determine the rate constants from experiments that measure individual decay-product concentrations. Practical application of such a theoretical framework usually requires assumptions that simplify the picture. One of the usual simplifications is consideration only of a single well-mixed space. Another is lack of differentiation of rate constants on the basis of aerosol size or chemical composition.

These simplifications aside, key issues of interest are the rates of attachment of the decay products to particles, as well as the rate as which free and attached decay products deposit on walls. (In many cases, deposition is parameterized in terms of the "deposition velocity," which is defined to be the ratio of the flux toward the surface to the concentration in the volume, which - in turn - equals the deposition rate times the volume-to-surface ratio for the space under consideration.)

By way of perspective, whereas typical ventilation rates are on the order of 0.5 h^{-1} and decay-product radioactive decay constants range from 1.6 h^{-1} for ^{214}Bi to 14 h^{-1} for ^{218}Po, rates of attachment to particles, for typical particle concentrations, appear to be on the order of 50 h^{-1}, with slightly lower rates - perhaps 15 h^{-1} - for plateout of unattached decay products onto interior surfaces. In contrast, rates for deposition onto walls of airborne particles and therefore of attached decay products are very low, on the order of 0.1 h^{-1}. Lowering the particle concentration tends to result in a higher overall rate of deposition onto the walls (because a higher proportion of the decay products are unattached) and a lower equilibrium factor. This condition can, of course, be attained by use of particle-cleaning devices, which also directly remove decay products from the air. However, the advantage indicated by the lower equilibrium factors and lower EEDC may be balanced by the fact that a higher fraction of the EEDC is associated with unattached decay products

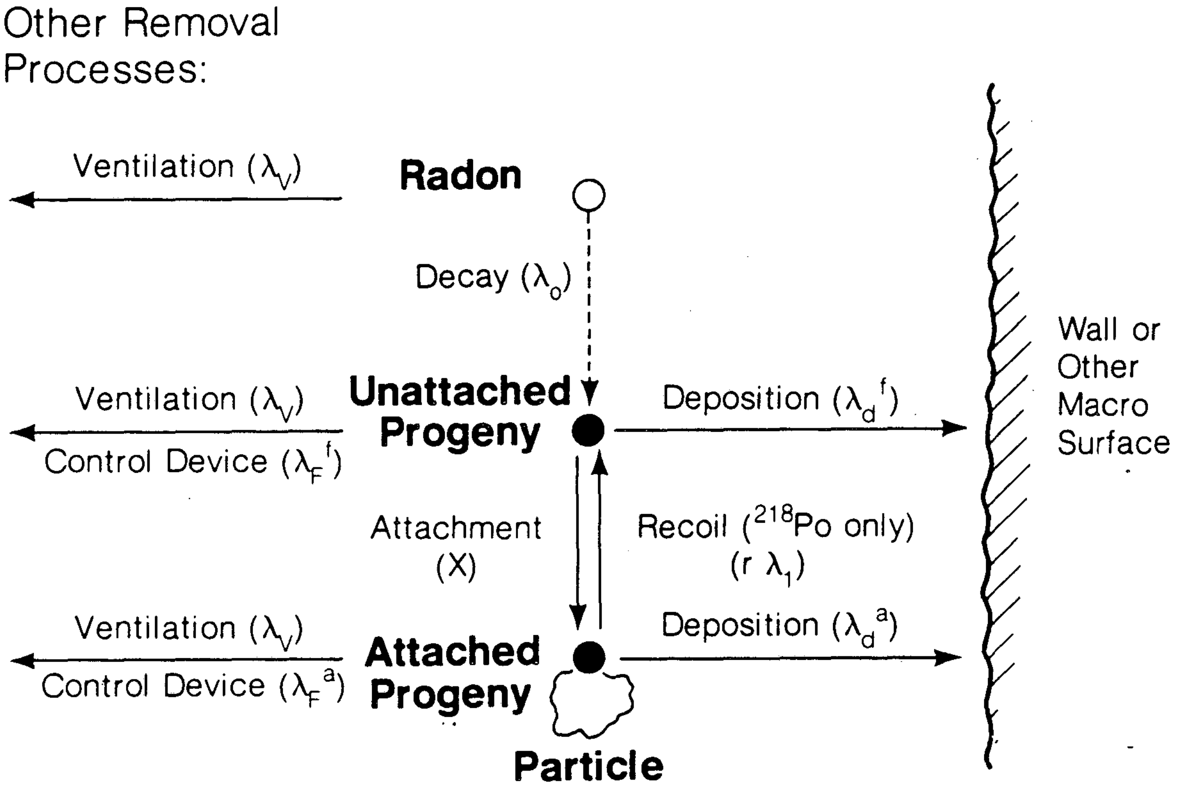

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Figure 6. Decay-Product Removal Mechanisms (Other than Radioactive Decay) and Associated Rate Constants

Once created by decay of its parent, a decay product (or "progeny") may attach to airborne particles, a process that is usually considered to be reversible for ^{218}Po because of the substantial recoil energy associated with alpha decay. Whether attached or not, a decay-product can be removed from the indoor air by plateout/deposition on indoor surfaces, by ventilation, by an air cleaning device, or by radioactive decay.

which appear to cause a more significant dose to the lung epithelium than the same amount of attached decay products.

(By way of comparison, the decay rate for ^{212}Pb, the most important ^{220}Rn decay product, is about 0.07 h^{-1}, so small that ventilation and other removal processes almost always lead to small equilibrium factors.)

B. Recent Results

The complexity and importance of radon decay-product behavior, as well as the potential interest in air-cleaning as a control techniques, has given rise to a substantial amount of work - both experimental and theoretical - on characterization of the decay products. Such work is reviewed in Knutson* and Phillips *et al.*,[⊕] as well as in Refs. 41-43. In this section, it is worth mentioning a few examples of important progress over the last several years.

Experiments in small and room-sized chambers, and related analysis in terms of a room-average model, have suggested the values for deposition rates indicated above.[44,45] This and earlier work - in particular, diffusion tube measurements[46] - has demonstrated that the rate at which "unattached" decay products plate out, while very high compared with particle deposition rates, is smaller than would occur if the decay products were present in the form of single unattached atoms (which would have a very high diffusion constant). The resulting conclusion that an unattached decay product is actually a cluster of atoms including a decay-product atom appears to be confirmed in experiments that measure the size distribution of decay products: they appear to divide into two regimes, one mode having a median diameter of about 100 nm, as might be expected based on the size distribution of particles typically present in a room, and a smaller fraction with median diameter in the vicinity of 10 nm, perhaps an order of magnitude greater than the size of a single ^{218}Po atom.[47]

Considering what is known about the behavior of radon decay products, estimates have also been made of the effect of air-cleaning devices on the radiation dose to the lung. Such estimates suggest that the radical reduction in decay-product concentration (given as EEDC or PAEC) that is possible by air cleaning may not cause a corresponding decrease in effective dose and in health effects; it is even possible that there is no decrease at all.[48] On the other hand, a detailed review of dosimetric models yields the result that, although the EEDC (or, equivalently, the PAEC) is an imperfect measure of dose and - ultimately - of potential health effects, it is still a reasonably good indicator, assuming that parameters are in the normal range.[49] These results would seem to suggest that - to the extent that air-cleaning devices result in particle concentrations outside the normal range - there is the potential that the EEDC (or PAEC) is no longer a good indicator of dose. Furthermore, more

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* E. O. Knutson, "Modeling Indoor Concentrations of Radon and Its Decay Products," Chapter 5 in Nazaroff and Nero.

[⊕] C. R. Phillips, A. Khan, and H. M. Y. Leung, "The Nature and Determination of the Unattached Fraction of Radon and Thoron Progeny," Chapter 6 in Nazaroff and Nero.

recent dosimetric studies suggest that, even for variability in decay product state caused by normal differences in particle concentration, the ^{222}Rn concentration may be a better indicator of effective dose equivalent than the EEDC (see James*).

Finally, Figure 6 does not explicitly indicate one of the potentially substantial influences on decay-product behavior, i.e., the fact that air within a room moves and that the pattern and rate of air movement can strongly affect deposition rates. Recent advances in modeling of air movement have made it possible to begin removing the simplifying assumption ordinarily used for simulation of decay-product behavior, that of a well-mixed room. The more detailed formulation permits treatment of the boundary layer more realistically, thereby providing a basis for determining the manner in which plateout rate (or deposition velocity) depends on conditions in the room, especially near the wall.[50]

The importance, not only of ventilation rate, but also of other aspects of air movement, indicates the need to consider in detail the manner in which buildings operate as a basis for understanding the behavior of radon decay products. A comparable conclusion arises in trying to understand how radon enters buildings, i.e., it has become clear that the building is not a passive object into which radon diffuses, but actively contributes to the entry of radon. And, indeed, both of these issues - radon entry and decay-product behavior - are linked to the question of ventilation and air movement in a more subtle way than was initially envisioned. The ventilation rate might be thought to influence indoor concentrations primarily by removing radon from the building interior and, to a lesser extent, by some reduction in the equilibrium factor. But we have found that the same factors that account for infiltration affect radon entry decisively and that comparable factors driving indoor air movement substantially affect decay-product behavior. Considering also the chemistry of the decay products, we see that properly understanding the behavior of radon and its decay products (as well as other pollutants) in buildings requires treatment of the problem as one of indoor atmospheric science, where airborne concentrations have a temporal and spatial dependence determined by sources and sinks, specific chemical and physical reactions, and complex indoor and outdoor wind fields.

VI. HEALTH EFFECTS

The ultimate reason for all the research on radon is the potential for exposures to radon decay products to cause ill effects among humans. A primary basis for this concern has been the increased incidence of lung cancer among mine workers exposed to higher-than-average levels of ^{222}Rn decay products, but several kinds of studies provide information on the effects of decay-product exposures. Together, these lead to some estimation (quantitative or otherwise) of the risk of lung cancer from indoor exposures and of the importance of this risk relative to other environmental insults.

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* A. C. James, "Lung Dosimetry," Chapter 7 in Nazaroff and Nero.

A. Types and Results of Health Studies

Studies of the effects of radiation generally fall into two categories. The first consists of epidemiological studies of humans exposed to radiation in the workplace, in the course of medical procedures, or in the environment, either natural or modified (the last including the extreme case of the Japanese nuclear bomb victims). The second class includes a variety of laboratory studies, including animal tests and cellular or physiological human studies. The principal classes of studies of the effects of radon decay products are epidemiological studies of miners and laboratory studies of animals, both subjected to relatively high exposures of ^{222}Rn decay products. In addition, a much broader range of studies, involving other types of radiation exposures and other types of biomedical investigations, contribute to a broader base of knowledge in the context of which the information specific to decay-product exposures is interpreted. A principal example is the recent effort to understand in detail the nature and site of the radiation doses resulting from inhalation of airborne decay products. The dosimetry of ^{222}Rn decay products, the evidence from human epidemiology, and the results from animal studies are discussed in detail in articles by James, Steinhausler, and Cross.* It is useful to note here the relationship among these research areas.

The miner studies offer a direct, albeit flawed, indication of the relationship between decay-product exposures and increased incidence of lung cancer. The principal study groups have been underground uranium miners in the United States, Czechoslovakia, and Canada, and iron, zinc, and lead miners in Sweden. The results of these studies, given in terms of increased risk per unit exposure or its equivalent, differ significantly among themselves, as indicated in Figure 7, where the incidence (following exposure) per unit exposure from various studies is plotted against the estimated cumulative exposure of the groups of workers examined in each study. The factors potentially contributing to differing values for dose-response factors are numerous, including differences in methods of analysis, inconsistency or error in the estimation of exposures, and differences in the presence of cofactors (such as diesel exhaust and ore dust), as well as the potential for a genuine dependence of the dose-response factor on cumulative exposure or on exposure rate.

Given the number of such factors, the apparent range in values is not surprising. Note that paying most attention to the data between about 100 WLM and 500 WLM (thereby neglecting the low-exposure data, which become less significant statistically as the exposure decreases, and the high-exposure end, where the risk per unit exposure appears to decrease), we are left with a range of roughly 5 to 35 lung cancers per year per WLM per million people. Taking an approximate median of 12, this means that for each WLM of total exposure an individual would suffer an attributable risk of 12×10^{-6} per year thereafter (or perhaps only after the latency period for appearance of lung cancer, this being one of the ambiguities of the studies). Given in terms of SI units, not often used in this context,

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* A. C. James, "Lung Dosimetry," Chapter 7, F. Steinhausler, "The Epidemiological Evidence for Health Risks," Chapter 8, and F. T. Cross, "Evidence from Animal Studies," Chapter 9 in Nazaroff and Nero.

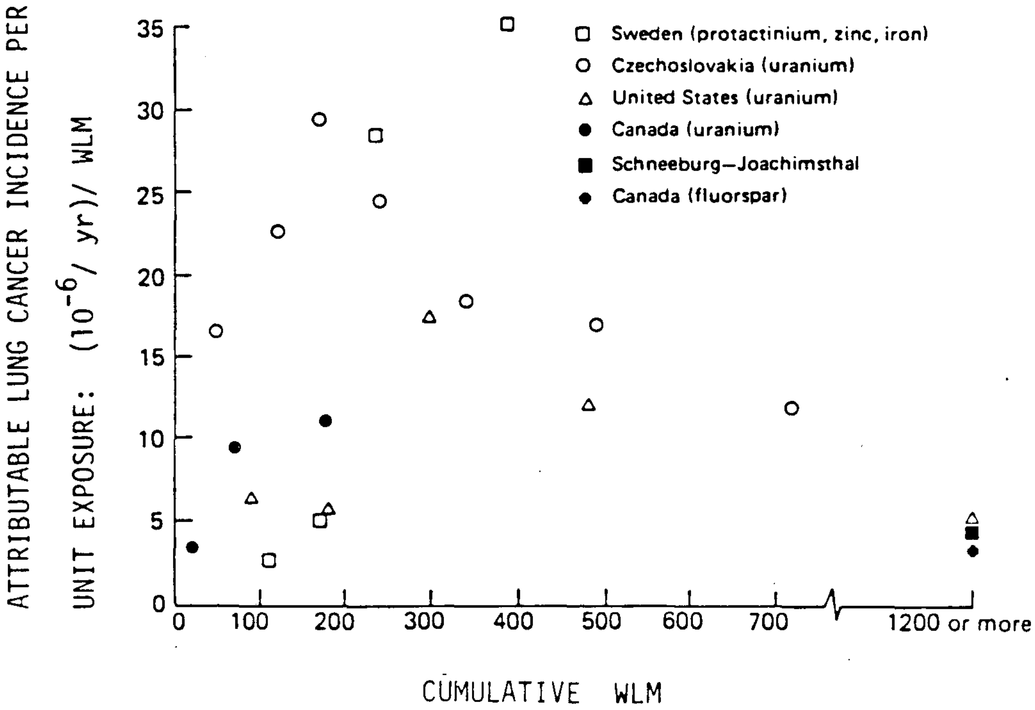

Figure 7. Attributable Lung Cancer Incidence per Unit Exposure versus Cumulative Exposure as Determined from Miner Studies

For most human data available, the figure shows the excess chance of cancer (in each year of followup) per unit exposure (in WLM) versus the cumulative exposure (in WLM) for each miner group. Adapted from Ref. 78.

the 12 x 10^{-6} $y^{-1}WLM^{-1}$ individual annual risk per unit exposure becomes (see appendix on quantities and units) 0.9 x 10^{-6} $y^{-1}(J\ m^{-3}s)^{-1}$. (For convenience, the combination J m^{-3}s is maintained as a unit of exposure.) Assuming, as is typically done,[51] that this attributable risk applies to a 30-year period of expression, this implies an individual lifetime risk per unit exposure of about 2.8 x 10^{-5} $(J\ m^{-3}s)^{-1}$ (or 3.6 x 10^{-4} WLM^{-1}). Various groups have considered the results from these studies, and - given the fact that they all utilize basically the same results - it is not surprising that they should arrive at similar values for the risk per unit exposure. The reader is referred, not only to Steinhausler, but also to a series of papers presented at the 1983 international conference at Anacapri.[49,52,53] Nonetheless, the "modest" differences in risk estimates - say a factor of 2 to 4 - can have substantial implications when considering the importance of radon exposures and strategies for controlling them.

Aside from the statistical uncertainties associated with the risk factor cited above, there are lingering underlying problems that have not been fully resolved. A major example is whether there is a synergism between decay-product exposures and smoking, or - put another way - whether the decay-product risk factor depends on smoking or smoking history. A related issue is whether a risk factor cited in "absolute" terms as above is better or worse than a "relative" risk factor, giving the attributable risk as a fraction of the "background" risk of lung cancer. In such terms, the rate of appearance of lung cancer appears to increase by about 1% per WLM exposure or 0.08% per J m^{-3}s.[53]

A major tool for examining such issues, and in general for permitting controlled experiments on factors affecting risk, is the use of animal studies wherein disease incidence or physiological change in groups of animals inhaling radon decay products is compared with rates in control groups. Cross treats these studies in detail, including major efforts using (Beagle) dogs, rats, and hamsters as subjects. Ref. 51 contains an earlier and useful summary of the animal studies.

The animal studies not only provide information on disease incidence as a function of exposure, but also provide a convenient means for investigating the detailed dosimetry of decay-product exposure, i.e., the manner in which the decay products deposit in the airways and the nature of the resulting radiation dose. Investigations of these matters are also carried out in humans. In fact, in recent years sufficient information has been accumulated about the structure and behavior of the human lung, as well as about the characteristics of airborne ^{222}Rn decay products, that it has become feasible to calculate the detailed pattern of deposition and exposure as a function of pulmonary characteristics and decay-product/particle mixture. James, as well as other references cited above, treat this question in detail.

A principal result of these detailed dosimetric studies is the ability to construct a rational framework within which to consider differing types of studies and to extend the results to conditions other than those actually present in the studies. For the present purpose, this offers two advantages: 1) it permits intercomparison of the results of studies of different types or undertaken under different conditions; and 2) it permits application of these results to exposures in the site of present interest, the indoor environment.

B. Risk from Indoor Exposures

Numerous investigators have recently examined the applicability of dose-response data from the miners studies to environmental, and in particular, indoor exposures. This requires consideration of a number of factors, including a population that is much broader in its constitution than the miner population, which tends to be healthy males in the prime of life. In addition, the miners were engaged in a higher level of physical activity during exposure, therefore with higher breathing rates, than the average level of the general population. Finally, the mine atmosphere contains a different mix of substances than an ordinary indoor environment, with the amount and size distribution of airborne particulates being of special interest.

A number of papers have examined such issues, as does James.* Most conclude that the dose-response factor derived from the miner studies can be applied relatively directly to environmental exposures. A relatively straightforward calculation indicates the size of those exposures: an average indoor ^{222}Rn concentration of, say, 50 Bq/m^3 implies, for an equilibrium factor of 0.4, an average decay-product concentration of 20 Bq/m^3. If two thirds of a 70-year lifetime is spent in this concentration, the lifetime exposure is 933 Bq m^{-3}y or (see appendix A on quantities and units) 164 J m^{-3}s (12.6 WLM). (If the other third of a life is spent at about 5 Bq/m^3 EEDC, a quarter of the indoor value, the total exposure is in the vicinity of 1000 Bq m^{-3}y, 180 J m^{-3}s, or 14 WLM.) Using the nominal lifetime risk per unit exposure given above, 2.8 x 10^{-5} (J m^{-3}s)$^{-1}$, this exposure implies an associated lifetime risk of 0.5% (neglecting detailed consideration of the population's age distribution, smoking habits, and other characteristics.) In point of fact, this risk (and risk factor) are slightly higher than that suggested by reviews under the auspices of national and international radiation protection organizations,[49,51] but somewhat lower than that used by the U.S. Environmental Protection Agency.[54] This risk, if applied uncritically to the U.S. population of 230 million, implies that 1 million of them might contract lung cancer as a result of ^{222}Rn decay-product exposures or that about 10-15% of the current annual incidence of 120,000 is due to this exposure (as compared with 14% implied by the relative risk factor cited above).

These estimates are quite uncertain, because of uncertainty in both the primary studies of the miners and the applicability of those results to environmental exposures of the general population. One aspect of this applicability issue is that the average exposure just cited, 1000 Bq m^{-3}y or 14 WLM, is well below the range where the epidemiological data have much statistical significance. On the other hand, this is not true of the higher levels found indoors. Lifetime exposure to an EEDC of 120 Bq/m^3 (associated with 300 Bq/m^3 ^{222}Rn) implies a total exposure of 5600 Bq m^{-3}y or 9.8 x 10^2 Jm^{-3}s (76 WLM). This exceeds exposures typical of

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* A. C. James, "Lung Dosimetry," Chapter 7 in Nazaroff and Nero.

present-day underground uranium miners and is in an exposure range where the associated lifetime risk, estimated to be 2-3%, assumes much more significance because there is no longer a need to extrapolate to low doses.

Regardless of the precise estimates, it is useful to put such risks in perspective relative to those from other environmental exposures. First, these exposures exceed those from other types of environmental radiation. Considering the population at high risk (e.g., in the million U.S. homes with ^{222}Rn concentrations of 300 Bq/m^3 or more), these exposures far exceed in significance even those accumulated in occupational settings, e.g., uranium mines, nuclear power plants, or medical facilities. Secondly, the average risk, whether 0.2 or 0.5%, far exceeds the levels at which other types of environmental exposures - such as those from outdoor airborne pollutants or toxic wastes - are considered to be significant. Regulatory investigation and even action often begins at lifetime risk levels in the range of 10^{-5} to 10^{-6}, a factor of a thousand below the estimated average risk from ^{222}Rn decay products, and even further below the 1+% risk of the heavily exposed portion of the population. This is especially ironic considering that, however uncertain the estimates of risk from decay-product exposures might be, they are much less uncertain than the estimates for other exposures, e.g., to potentially carcinogenic chemicals. The estimates for such chemicals are typically based on animal tests at exposures thousands of times higher than human exposures, with assumptions that are purposely selected to overestimate the risk in order to be conservative in protecting the public. A further irony is that concentrations of such chemicals may be much higher in the indoor environment than outdoors, where the main regulatory attention is focussed.

VII. IDENTIFYING AND CONTROLLING INDOOR CONCENTRATIONS

Given the apparent importance of indoor exposures to ^{222}Rn decay products, an immediate question is how to assure that the levels to which the population is exposed are acceptable. The answer to this question entails several related issues, including the narrow ones of measurement and control techniques and the broader problem of formulating a control strategy.

A. Measurement Techniques

Measurement of the airborne concentration of radon or its decay products is based on detection of the radiation associated with radioactive decay. This radiation includes, not only the alpha and beta radiation indicated in Figure 1, but X or gamma radiation as the immediate product of decay from the "excited" state in which it is left to its "ground" or unexcited state. Any of these forms of radiation is capable of creating ions in materials through which it passes, and measurement methods are based on detection of this ionization or its aftermath.

In spite of this uniformity in the basic principle of detection, practical measurement techniques take widely different forms, substantially affecting their range of application. Specific techniques are designed for measuring the concentration of radon, the EEDC, or - indeed - the concentration of individual decay products. Some techniques are suitable for short-term measurements, where a "grab" sample of air is taken, or for continuous real-time measurements, where repeated samples are taken. Other techniques are designed for intermediate to long-term sampling, where only the average concentration is sought. Some techniques may be self-contained, requiring the presence of a moderate to large piece of equipment at the site of measurement, while other techniques deploy a small sampler or detector, which must be returned to a central analytical laboratory for readout. Finally, some techniques are inexpensive or easy to use, while others are suitable for use only in the context of experiments carried out by experienced scientists.

Measurement techniques are treated in a number of specific instrumentation reviews.[55,56] As an illustration of techniques, however, it is worth mentioning three widely used techniques that are suitable for use in monitoring large numbers of houses:

1. Grab-sample monitoring for either ^{222}Rn or its decay products can be performed using highly-portable self-contained instruments that use a scintillation material to measure alpha particles emitted in a small collection cell (for ^{222}Rn) or from a filter (for the decay products). The entire measurement is completed within some tens of minutes, therefore only giving the concentration at a specific time. Considering the potential variability of concentration with time, this does not give a direct measure of the average exposure rate in a home.

2. A charcoal sampler with dimensions on the order of 10 cm can be used to collect ^{222}Rn over a period of a day to a week, then sealed and returned to a laboratory, where the gamma radiation from decay products contained in the sampler can be measured.[9] This technique is not suitable for more than one week because late in this period much of the ^{222}Rn collected at the beginning will be gone because of its 3.8-day half life.

3. An etched-track detector with dimensions of only about 3-4 cm can be placed in a home for periods from weeks to a year, then returned to a laboratory where the plastic detector material is etched to show tracks left by the passage of alpha particles from the decay of ^{222}Rn and its decay products.[8] As ordinarily deployed, such detectors have a filter permitting entry of ^{222}Rn, but not its decay products (for which it is difficult to calibrate the device). This techniques has the advantage of giving a long-term average concentration, which - however - might be considered a disadvantage if a quick result is desired.

In the context of a program for identifying and reducing excessive concentrations, the suitability of each of these techniques would depend on the objective of any particular measurement.

B. Control Techniques

The techniques available for controlling indoor concentrations of ^{222}Rn and its decay products correspond quite closely to the basic factors found to affect concentrations. For several of the major classes of indoor pollution - whether radon, combustion products, or airborne chemicals - these factors include the source strengths (or entry rates) for the pollutants of interest, the ventilation rate (and pattern), and reactions of the pollutants with each other and with the building or its contents. For each pollutant class, concentrations appear to be distributed approximately lognormally, and the largest contributor to the width of the distribution is typically the source strength, but - in each case - variability in ventilation rates and in reaction rates contribute significantly. For the case of radon, as for the other pollutants, it appears that control techniques altering the entry rate, discussed in Scott,* offer greater potential reduction than other techniques.

The strong variability found in radon entry rates suggests the efficacy of a control program aimed at minimizing entry rates, particularly where they are large. Considering the importance of pressure-driven flow of soil gas into houses through their understructures, substantial attention has been given in recent years to the potential for reducing this flow. It is clear that the use of better barriers, sealants, and construction techniques can have a significant effect on the radon entry rate, but this potential appears limited in most cases, one reason being that a substantial pressure-driven entry rate can persist if only a few passages for soil gas remain or if new imperfections appear as a result of slight movement of the house understructure.

An alternative approach that has the potential for reduction of entry rates by large factors is to apply a technique that reduces (or, sometimes, increases) the air pressure immediately under the house, thereby disrupting the flow of air that carries radon from underlying soil into the house. This might be thought of as offering an alternative route for the radon flux from the ground or, alternatively, as creating a pressure barrier. And, indeed, in certain cases, where the main entry route is highly localized, as through a drain tile and sump system, what one does is provides local venting to the outdoors. In the more general case of a basement (or slab-on-grade), one or a few pipes with a small fan to depressurize (or pressurize) the material (preferably gravel) immediately below the concrete floor can greatly reduce the radon entry rate. In the case of crawl spaces, active ventilation of the space below the house can easily be accomplished, although - as noted above - careful sealing of the floor may be quite practical in this case.

For situations where large entry rates are responsible for excessive indoor concentration, such entry reduction techniques appear ordinarily to have the greatest potential effect. However, there are also circumstances where increases in ventilation rate are appropriate: where the ventilation rate in question is unusually low, where source reduction techniques do not appear effective for the case at hand (e.g., if the source is unusual building materials incorporated into the structure), or where - as in rare cases of extremely high concentrations - an immediate, if only temporary, solution is required. The primary limitation of increased ventilation rates, especially in homes with very high concentrations, is that reduction of indoor concentrations by large factors will require increases in the ventilation rate by large factors. This is often impractical, uncomfortable, or too expensive, at least for the long term. For homes where only modest reductions are sought, ventilation rate increases are

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* A. G. Scott, "Preventing Radon Entry," Chapter 10 in Nazaroff and Nero.

quite practical, including use of mechanical systems that recover energy that would otherwise be lost - either by incorporation of an air-to-air heat exchanger between incoming and outgoing air streams or by simple recovery of heat from exhaust air. (However, an exhaust ventilation system has to be used with some care, since it may result in increased depressurization of the house and larger radon entry rates.)

An alternative means of control is, of course, use of air cleaning systems to remove the decay products, discussed in detail elsewhere.[*] The most common of these employ particle removal techniques such as filtration and electrostatic precipitation. However, although the better devices using such techniques can substantially reduce the decay-product concentrations as measured by EEDC or PAEC, their effect on the dose to the lung is far from clear. As a result, it is generally thought that source-strength reduction is the best control technique, with the next best option being increased ventilation rates.

C. Strategies

Knowledge of measurement techniques and methods for control does not constitute a strategy for identifying and controlling excessive concentrations. Identification and control can be achieved only in a more general framework that embodies the objectives to be met and the overall strategy for achieving them. In this context, specific measurement and control techniques are merely the tactics or tools for a campaign against the radon problem. The fundamental elements of a control strategy are discussed in more detail in Nero,[⊕] but it is worth emphasizing here that two of the elements are mainly technical and two are mainly questions of valuation and responsibility. The technical elements are 1) developing the actual scheme for identifying areas and individual houses with high concentrations, and 2) formulation of a logical structure for choosing what control technique(s) should be attempted in each type of situation. Just as important are 3) that some agreement be reached on the objectives of the strategy, usually embodied in a structure of concentration limits (or guidelines) or comparable standards, and 4) that responsibility (both scientific and fiscal) be allocated for locating houses with excessive levels and for implementing the appropriate control measures. Improvements in scientific understanding and technical capabilities are prerequisites for coping with the problem of indoor radon. But little can be done effectively without carefully framing various objectives and creating an overall strategy suited to achieving them.

VIII. APPENDICES

A. RADIOACTIVE QUANTITIES AND UNITS OF MEASURE

[+] N. Jonassen and J. P. McLaughlin, "Radon Progeny Removal," Chapter 11 in Nazaroff and Nero.

[⊕] A. V. Nero, "Elements of a Strategy for Control of Indoor Radon," Chapter 12 of Nazaroff and Nero.

Amounts of radioactive material can be specified in principle by either mass or activity, the latter being more conventional. The activity is the actual rate at which atoms decay radioactively, and the standard international (SI) unit for activity is the Becquerel (Bq), equal to a decay rate of one per second (s^{-1}). This unit (or its traditional equivalent, the Curie) is adequate for expressing activity and (activity) concentration, not only for radon, but also for its decay products and any combination of them. The present work utilizes SI units primarily, but traditional units are defined below in terms of the modern units. In the text, the older units are sometimes given parenthetically or when citing work that is difficult to translate without modifying sense or quantitative results.

For decay products, the collective quantity of most use is the equilibrium-equivalent decay-product concentration (EEDC). Based on the alpha decay energies and half lives of the ^{222}Rn decay series (see Figure 2), the EEDC is given in terms of the individual decay-product concentrations as:

$$\text{EEDC} = 0.104 \times I(^{218}\text{Po}) + 0.514 \times I(^{214}\text{Pb}) + 0.382 \times I(^{214}\text{Bi}),$$

with concentrations of each decay product (and of the EEDC) given in Bq/m^3 or pCi/l, as indicated in Table 4. (The ^{214}Po concentration does not contribute significantly to this expression because of its very short half life, causing very few ^{214}Po atoms to be present in air as compared with the other short-lived decay products. However, the ^{214}Po alpha energy is the largest contributor to the coefficients in this expression, since the presence of each of the previous three decay products implies a ^{214}Po decay.) The analogous expression for the ^{220}Rn series (cf. data in appendix D, below) is:

$$\text{EEDC} = 0.913 \times I(^{212}\text{Pb}) + 0.087 \times I(^{212}\text{Bi}).$$

(^{216}Po and ^{212}Po do not contribute directly to this expression - again, because of their very short half lives - even though ^{212}Po contributes the dominant alpha energy.)

Historically, the potential alpha energy concentration (PAEC) itself was used as the measure of decay-product concentration, with the standard unit being working level (WL). The corresponding SI unit is that of energy per unit volume, J/m^3, and: PAEC = EEDC x 5.56×10^{-9} J/Bq (or 1284 MeV/pCi) for the ^{222}Rn series. (The corresponding factor for the ^{220}Rn series is 7.56×10^{-8} J/Bq or 1.75×10^4 MeV/pCi.) In any case, the PAEC is effectively supplanted by the EEDC (i.e., the decay-product concentration itself), except possibly in specifying exposures. (Note that in one respect the PAEC is a more effective measure of concentration than the EEDC, i.e., in comparing ^{222}Rn and ^{220}Rn: per unit radioactivity, the ^{220}Rn decay products carry 13.6 times as much PAEC as the ^{222}Rn decay products.)

The basic measure of exposure is essentially the product of concentration (or its equivalent) and time. EEDC x time has units of Bq m^{-3}s, although hour, month, or year might also be used as the unit of time. PAEC x time has SI units of J m^{-3}s, but the traditional units are working-level hour (WLH) or - more frequently - working-level month (WLM),

Table 4. SI Units and Equivalents for Traditional Units[a]

Parameter, SI Unit	Conversion for Traditional Unit
Activity, Bq	1 Ci = 3.7×10^{10} Bq (1 pCi = 0.037 Bq)
Concentration, Bq m^{-3}	1 pCi/l = 37 Bq m^{-3}
PAEC, J m^{-3}	1 WL = 1.3×10^{5} MeV/l = 2.08×10^{-5} J m^{-3}
$EEDC_{222}$, Bq m^{-3}	1 WL (PAEC) = 3740 Bq m^{-3}
$EEDC_{220}$, Bq m^{-3}	1 WL (PAEC) = 275 Bq m^{-3}
Exposure, J m^{-3}s	1 WLM = 12.95 J m^{-3}s
Exposure, Bq m^{-3}y	1 WLM = 73.9 Bq m^{-3}y
Exposure rate, J m^{-3}	1 WLM/y = 4.11×10^{-7} J m^{-3}
Exposure rate, Bq m^{-3}	1 WLM/y = 73.9 Bq m^{-3}

[a] The data on which these conversions and the expressions for EEDC are based are taken from reference 57.

equal to 173 WLH. Exposure rate, in the SI units, simply reduces to average EEDC (or alternatively PAEC), but in the old units is usually WLM/y.

B. VENTILATION RATES: DISTRIBUTION AND DEPENDENCE

A key factor affecting indoor pollutant concentrations is the ventilation rate. Characterizing ventilation rates in building can be difficult, since many buildings are complex structures, and paths and rates of air movement can depend substantially on location and time. Even considering only homes, the question is complicated, both because of the wide variety of housing structures and because - even for a given (and even simple) structure type - the pattern of air movement is complex. These complexities have substantial implications for the behavior of any reactive species, including radon decay products, and for the manner and degree of radon entry, as described elsewhere in this chapter.

Nonetheless, restricting attention to homes - consisting of relatively self-contained living units - simplifies the picture sufficiently that the ventilation rate has a less ambiguous meaning (although even this can be complicated in the case of multi-apartment buildings in which air is recirculated among different units). Moreover, the rate can actually be measured relatively directly. For example, the ventilation rate measurements contributing to Figure 3 were performed by injecting a tracer gas, sulfur hexafluoride (SF_6), into each house, then - after a mixing time - measuring the concentration as a function of time using an infrared analyzer: the ventilation rate equals the rate of decay of the tracer concentration. And in a current passive measurement technique, a tracer is released from a small source at a constant rate and a collecting monitor, consisting of a diffusion tube and an absorber (characteristic of a number of techniques for passive sampling of gaseous pollutants indoors), measures the average concentration during the time this system is deployed:[58] this measured value is then proportional, in first order, to the average of the inverse of the ventilation rate, $I_{tracer} = S_v(1/\lambda_v)_{average}$.

The results of such measurements confirm the expectation that ventilation rates vary substantially from one country to another, from one class of buildings to another, and even within the same general structural class, e.g., single-family houses or multi-unit apartment buildings. Still, for most homes in the United States (and many industrialized nations), the average ventilation rate during seasons when the windows are kept mostly closed lies in the range 0.1-1 h^{-1}. This total ventilation rate is made up of three components: infiltration of air through small openings or imperfections in the building shell, exchange of air through windows or doors that are partially or temporarily open, and ventilation supplied mechanically by exhaust fans or other systems. Each of these components varies with time, not only from one season to another, but also from one day to the next, and even over shorter periods. As a result, the total ventilation rate has a significant time dependence, even within a single building. Even during the heating season, when - for many homes - infiltration supplies most of the ventilation, substantial variability occurs, as noted below.

It is important, in considering the results of ventilation rate measurements, to pay attention to what component of the ventilation the monitoring protocol was designed to measure. For example, the basic tracer injection technique mentioned above is often employed with windows closed, therefore measuring only the infiltration rate, often the dominant winter component. In contrast, the integrating passive technique measures the average inverse total ventilation rate.

Like the radon concentrations discussed in this chapter, measurements of infiltration rate in any sample of buildings are often found to be distributed lognormally, i.e., the distribution of the log of the ventilation rate is approximately "normal" or Gaussian. An example of this is a set of measurements taken in 200 single-family houses in several cities of the United States, where weatherization programs were taking place.[59] The average infiltration rate was found to be 1.1 h^{-1}, and a more recent determination of the geometric mean and standard deviation yields 0.90 and 2.13, respectively, as discussed elsewhere.* Considering the age and condition of these houses, their infiltration rates are higher than the average U.S. value, which in turn is significantly higher than in northern European countries where substantial energy-saving efforts appear to have effected significant reduction in infiltration rates.

In recent years, the processes driving infiltration rates in homes have come to be better understood, and even embodied in relatively simple quantitative models. Basically, we now know that infiltration occurs because of air convection into and out of the house, driven by small pressures across the building shell arising from two factors: winds, which obviously exert small - albeit complex - forces on the building and its surroundings (specifically, the ground), and temperature differences between indoors and outdoors. During the heating season, this temperature difference causes a "stack" effect (much as in a fireplace and chimney), such that air is drawn in near the bottom of the structure and forced out toward the top. The pressure differences caused by winds and temperature differences are roughly comparable in size, averaging on the order of a few pascal (with higher values in relatively severe climates). Because the size and pattern of the associated pressures vary markedly from one time to another, it is not surprising that infiltration rates also vary substantially.

A number of groups have formulated models giving the infiltration rates in terms of the pertinent environmental parameters, and their success is reviewed in a recent publication of the Air Infiltration Centre.[60] As successful as any model - even complex ones - is a simple parameterization of the infiltration rate in terms of the wind speed V, the temperature difference dT, and an effective "leakage area" A_0. In these terms, the infiltration component of the ventilation rate may be expressed as

$$v_i = A_0(f_w V + f_s dT^{1/2});$$

here f_w is a parameter accounting for local and terrain shielding effects,

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* W. W. Nazaroff *et al.*, "Water," Chapter 4 in Nazaroff and Nero.

the distribution of leakage area around the building envelop, and the height of the building relative to the height at which the wind speed is measured; f_s is a stack parameter accounting for the building height and the distribution of leakage area.[61] Using this model together with applicable meteorological information, heating-season infiltration rates have been estimated from measurements performed in 200 houses distributed throughout the United States and Canada. The average was found to be 0.67 h^{-1} (with a significantly lower value, 0.48 h^{-1}, found for houses less than two years old at the time of measurement).

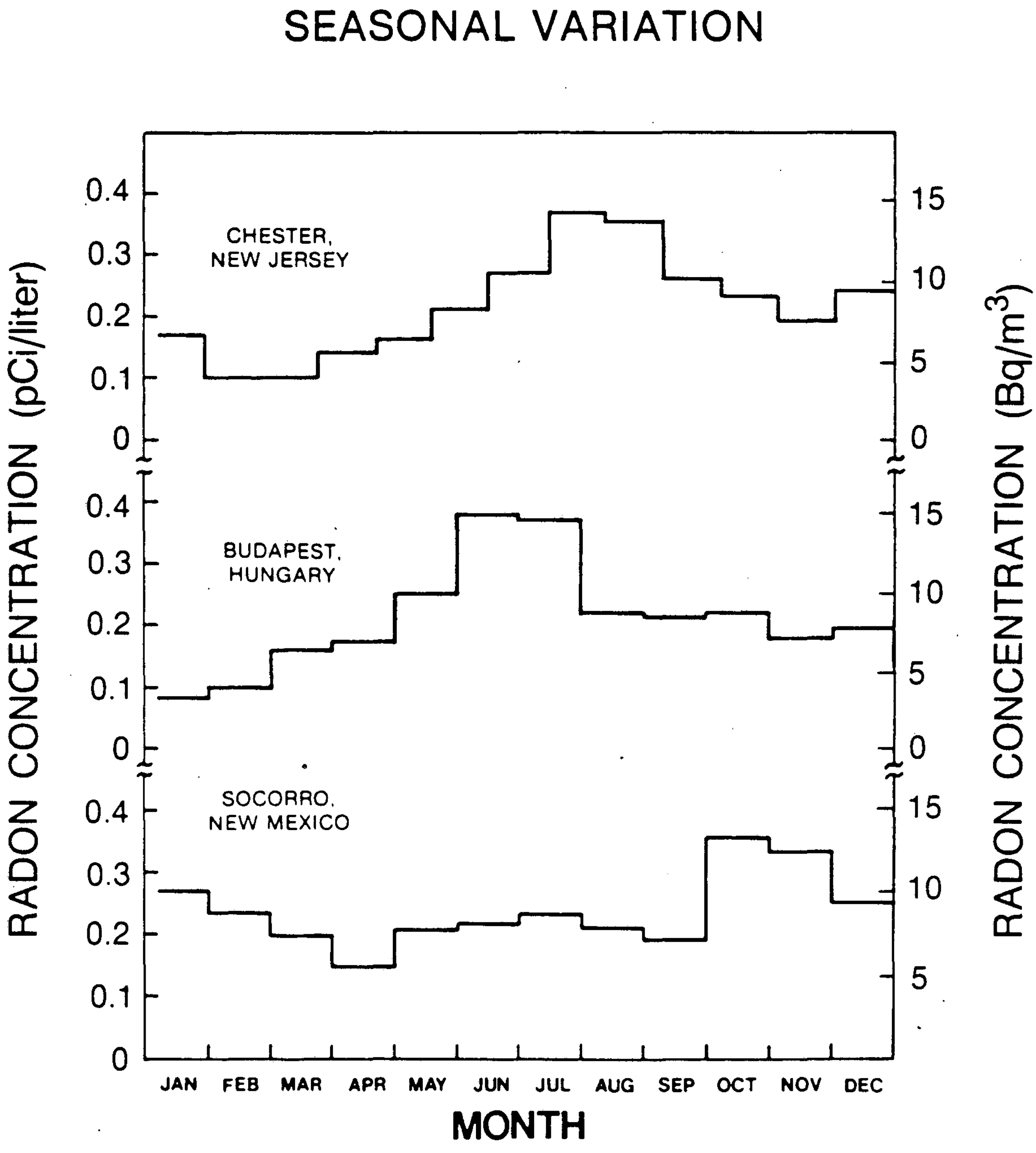
C. Outdoor Concentrations of Radon and Its Decay Products

During the last two or three decades, a significant effort has been devoted to monitoring of outdoor radon concentrations, either as part of the overall characterization of environmental radiation exposures or to determine environmental radon levels as a contrast to those in occupational or indoor settings. A paper by Gesell reviews and analyzes the results from such efforts, examining the variability of outdoor ^{222}Rn concentrations with time of day, season, altitude, and location.[62]

The magnitude and behavior of outdoor concentrations is indicated by Figure 8, where continuous measurements taken at three locations are seen to yield concentrations (averaged over approximately one month) in the range of 4 to 15 Bq/m^3 (0.1 - 0.4 pCi/l), with a significant seasonal variation whose behavior differed among the locations. Examining the same data versus time of day yields average concentrations having a substantial diurnal variability (roughly a factor-of-two range in each case), with the lowest values occurring at noon or soon thereafter and the highest in the middle of the night or early morning. The observed variation with time can be attributed to differences in environmental conditions, affecting not only the movement of radon out of the ground, but also - and probably more importantly - the rapidity of mixing in the atmosphere. This mixing determines the height dependence of concentration, which varies only modestly over distances comparable to the height of buildings (e.g., about a factor of 2 from 1 to 100 m or from 0.01 to 1 m height above the ground).

Significantly higher and lower values than found in Figure 8 have been found at other locations, e.g., averages of 28 Bq/m^3 in Grand Junction and 0.5-1 Bq/m^3 in Alaska; low values also occur in areas substantially influenced by marine air. Moreover, instantaneous values can also vary markedly from the averages. Gesell suggests that the mean value for the contiguous United States lies between 0.1 and 0.4 pCi/l (4-15 Bq/m^3), probably in the vicinity of 0.25 pCi/l (10 Bq/m^3). Data from other countries are not dissimilar.

Some information is also available on the decay products in outdoor air, where - because of the relative absence of surfaces (compared with indoors) - one might expect higher equilibrium ratios. For example, measurements in New York yielded ratios around 0.45 (with uncombined fraction of ^{218}Po of 9%) at a rural location and 0.40 (uncombined fraction Of 5%) on a city sidewalk.[63] Subsequent measurements outside eight houses in New York and New Jersey gave mean equilibrium ratios averaging 0.79, considerably higher than the indoor ratios measured.[5] More recent data (cf.



XBL 871-9911

Figure 8. Year-Long Variability of Atmospheric ^{222}Rn Concentration

Measurements were performed 1 m above the ground in three different locations (figure adapted from Ref. 62, by permission).

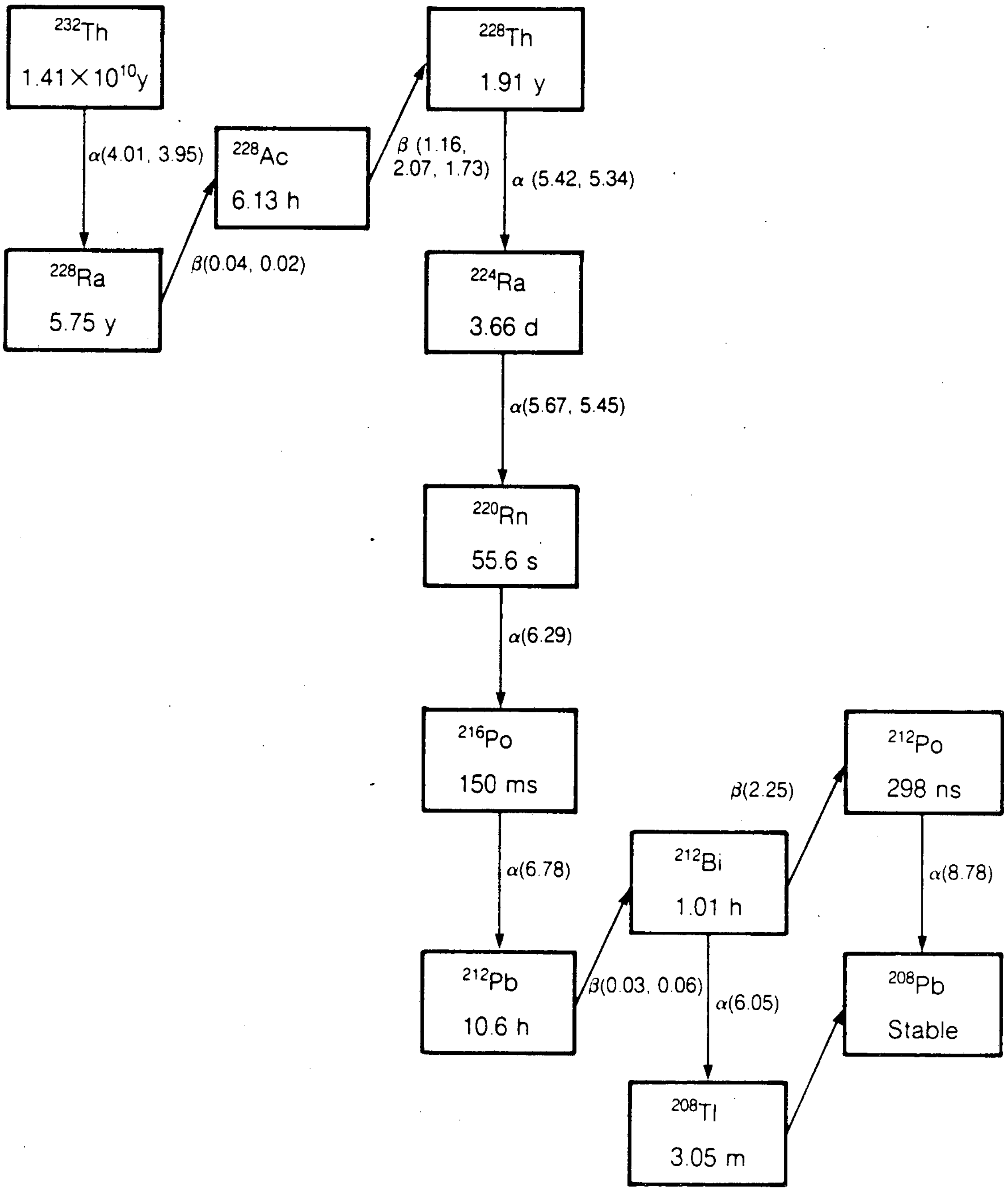


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Figure 9. Thorium 232 Decay Chain, including ^{220}Rn and its Decay Products

Radon 220 and its parent, ^{224}Ra, are members of the ^{232}Th decay series. Airborne concentrations of ^{212}Pb and ^{212}Bi are of prime radiological interest due to their potential for retention in the lung, leading to subsequent irradiation by the alpha decay of ^{212}Po. Cf. Figure 2 for ^{238}U decay chain; half lives and energies taken from E. Browne and R. Firestone, *Table of Radioactive Isotopes* (V. S. Shirley, Ed.), Wiley, New York, 1986.

Table 5) yields an equilibrium factor of 0.56.[64]

A ^{222}Rn concentration of 10 Bq/m^3, while completely negligible compared with the higher indoor values observed, is still significant compared with typical indoor concentrations. Therefore the outdoor contribution has to be considered in making precise estimates of human exposures to ^{222}Rn decay products. It also has to be considered in examining the sources of radon found indoors: the approximately 10 Bq/m^3 coming in with outdoor air is approximately 20% of the average ^{222}Rn concentration observed in U.S. single-family homes and an even larger percentage of concentrations in other types of buildings.

D. Radon 220 and Its Decay Products

Our present understanding of the occurrence and behavior of ^{220}Rn and its decay products is very limited compared with our knowledge of the ^{222}Rn series. Not nearly as much effort has been devoted to ^{220}Rn characterization, because preliminary and modest evidence - both theoretical and experimental - suggests that airborne concentrations of ^{220}Rn and its decay products are not as important from a radiological point of view as their more common relatives. This occurs in spite of the fact that the ^{232}Th decay series, shown in Figure 9, has approximately the same (activity) concentration in the earth's crust as the ^{238}U series, about 25 Bq/kg.[65,66]

The differences have to do partly with the short half life of ^{220}Rn, 1 min as compared with 4 d for ^{222}Rn, and partly with the details of radon and decay product behavior in the indoor or outdoor environment and inside the lung. The first factor sharply limits the time available for transport from the source material, so that only a small thickness of building material or soil serves as an effective source for ^{220}Rn entering the atmospheric environment, indoors or outdoors. The second affects the concentrations present in the air and the dose delivered by the decay products that deposit in the lung. The net result of all these factors is that - at least for indoor ^{222}Rn concentrations in the normal range - the ^{220}Rn decay products (in particular ^{212}Pb) appear - on the basis of limited evidence - to have a potential alpha energy concentration (PAEC) that is less than, but still significant compared with, that from the ^{222}Rn series. However, the PAEC itself does not reflect the present understanding that a given amount of alpha energy from the ^{222}Rn series has substantially more biological impact than alpha energy from the ^{220}Rn series (cf. James*).

We consider first the concentrations expected if transport from the source to the air were dominated by diffusion for both of these radon isotopes, as is likely to be the case outdoors. The depth of soil serving as a source is then indicated by the diffusion length, which is proportional to the square root of the half life. The ratio of the ^{220}Rn to ^{222}Rn half lives is 1/5940 (i.e., 55.6 s/3.82 d), yielding 1/77 as the diffusion length ratio. This indicates the relative depth cleared per

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* A. C. James, "Lung Dosimetry," Chapter 7 in Nazaroff and Nero.

characteristic time, which is given by the half lives. Since the activity concentration in the soil is similar for the ^{232}Th and ^{238}U series, the relative rate at which ^{220}Rn and ^{222}Rn activity escape from the surface of the source material is the relative thickness cleared (1/77) divided by the relative time scale (1/5940), yielding a factor of 77. Thus much more ^{220}Rn activity is expected to escape than ^{222}Rn activity! (However, because of the difference in half lives, this greater activity is supported by only 1/77th as many atoms.) In contrast, it is easy to show that - if the transport mechanism is pressure-driven flow of air through the source material and into the open air - the rate at which activity leaves the surface is the <u>same</u> for the two isotopes. (In this case, only 1/5940th as many ^{220}Rn atoms escape.)

For a typical ventilation rate of 1 h^{-1}, the assumption of pure diffusion (with no barriers between the source and the air) can therefore be shown, using Eg. 1, to imply about twice as much ^{220}Rn activity in indoor air as ^{222}Rn. As we will see, the observed ratio is not usually this large. However, for outdoor air, the correspondence is reasonably good. For example, a review by Schery[64] of ^{220}Rn and its decay products indoors cites typical fluxes from open soil of 2 Bq m^{-2} s^{-1} for ^{220}Rn and 0.016 Bq m^{-2} s^{-1} for ^{222}Rn, a ratio that is only slightly larger than the factor of 77 suggested above. Based on a simple eddy diffusion model to represent the transport of radon upward into the atmosphere, he concludes that the concentrations of ^{220}Rn and ^{222}Rn 1 m above the ground ought to be 9 and 6 Bq/m^3, respectively, compared with his (limited) measurement results of 16 and 6 Bq/m^3, respectively. He notes that his tentative result of 4.6 nJ/m^3 for $PAEC_{212}$ (the PAEC from ^{212}Pb, the primary ^{220}Rn decay product: see appendix A on quantities and units), versus 20 nJ/m^3 for $PAEC_{222}$ (the PAEC from the ^{222}Rn decay products), compare reasonably with more extensive measurements made in the Federal Republic of Germany.[67] (These results imply an equilibrium factor for ^{220}Rn decay products that is extremely small - not surprising considering the relative half lives of ^{220}Rn and ^{212}Pb - and an equilibrium factor fo ^{222}Rn decay products greater than 0.5.)

The data on indoor concentrations are limited and more difficult to interpret. A number of workers cite results for the ratio of $PAEC_{220}$ to $PAEC_{222}$ centering around 0.5. See, for example, European data ranging from 0.3 to 0.8, summarized in ref.68 and North American results from Elliott Lake, Canada, of 0.3[69] and for the United States of 0.6.[70] These results indicate the potential significance of the ^{220}Rn series indoors. However, our understanding of the factors affecting the ^{220}Rn concentrations, both absolutely and relative to ^{222}Rn, is tentative at best.

For example, if entry of both radon isotopes is diffusion dominated, as has been supposed in some classes of European housing (particularly apartment structures), and concentrations in the source materials are similar, then indoor radon concentrations ought to be similar for the two isotopes (as noted earlier), the activity concentration of the decay products ought to be a factor of 5 or so less for ^{220}Rn than for ^{222}Rn (because the long half life of ^{212}Pb causes most of it to be removed by ventilation or deposition rather than by decay), and the $PAEC_{220}$ ought to be greater than the $PAEC_{222}$ by a modest factor, e.g., 2 or 3. Even in European housing the ratio is closer to 0.5, suggesting significant differences in the source concentrations or generation parameters for the two isotopes, barriers to the presumed diffusion process, other mechanisms for radon entry, or unaccounted-for differences in the decay-product behavior.

One difficulty is obvious, i.e., at the concentrations typical of European apartment structures, the ^{222}Rn entering the indoor air from structural materials can almost be equaled by that entering from the outdoor air. In contrast, the ^{220}Rn half life is so short compared with ventilation time constants that no comparable contribution arises for this isotope.

Another factor to be considered is the more interesting one that pressure-driven flow has been shown to be the dominant contributor to indoor ^{222}Rn for many types of structures and may also be significant for ^{220}Rn. For homes such as those that dominate the U.S. housing stock, little diffusion from the floor and wall material is expected in any case (because of the use of wood and the painting of other types of materials), so that some other entry route is likely to dominate observed concentrations of the ^{220}Rn series. As indicated above, comparable concentrations in the source materials would imply similar activity entry rates for a pure flow process. Because of the relative size of decay rates and typical ventilation rates, this would imply ^{220}Rn concentrations only about 2% of ^{222}Rn concentrations (and, using the same argument as above) a similar ratio of PAECs. Thus the concentration ratios expected differ by about two orders of magnitude, depending on whether diffusion or flow is the presumed mechanism, and the actual observations - in both European and American housing - lies approximately at the geometric mean. This suggests the dominance of different entry mechanisms for the two isotopes or a combination of diffusion and flow applying to both, a possibility that was indicated in the early work on pressure-driven flow.[27]

However, the measurements of Schery in U.S. buildings[70] also afford the possibility for an interesting comparison, indicated in Table 5. There are indicated the results of outdoor measurements cited above, as well as the average indoor $PAEC_{212}$ and $PAEC_{222}$ from Schery's 68 measurements in buildings in 21 states. The detailed data show a significant correlation between these two parameters (in contrast to the work of Ref. 69), suggesting the possibility of common entry mechanisms (or, of course, common removal mechanisms). Also shown are estimates of the indoor radon concentrations, based on the $PAEC_{222}$ for ^{222}Rn and on one of Schery's illustrations for the ^{220}Rn. These concentrations are in good agreement with his average ^{220}Rn/^{222}Rn ratio of 0.23, which is - consistent with the comments above - at the (geometric) midpoint between the expectations for purely diffusion and purely flow entry mechanisms.

However, the ^{222}Rn concentration estimate is considerably less than the average concentration in U.S. single-family homes suggested on the basis of the studies discussed in the body of this chapter, i.e., 55 Bq/m^3. This may occur because of the season during which Schery's measurements were made, roughly April through September in northern climes and October through March in southern.[70] In any case, these results do not appear to adequately represent the higher ^{222}Rn concentrations observed. Very few of the results even exceed the <u>average</u> concentrations found in U.S. single-family homes.

Furthermore, it is interesting to note the relative sizes of the

Table 5. Comparative Concentrations of ^{220}Rn and ^{222}Rn and Their Decay Products[a]

	Outdoor (1m height) Rn (Bq/m^3)	 PAEC (nJ/m^3)	Indoor Rn (Bq/m^3)	 PAEC (nJ/m^3)
^{220}Rn Series	16	5[b]	3-4[c]	21[b]
^{222}Rn Series	6	20	15-20[d]	44

[a] Based on refs. 64, 70.
[b] These are only the PAEC's for ^{212}Pb, the dominant contributor of the ^{220}Rn decay products.
[c] Estimated from Fig. 3 in ref. 70.
[d] Estimated from PAEC (44 nJ/m^3) assuming an equilibrium factor of 0.4-0.5.

PAECs cited in the table. The indoor PAEC$_{222}$ is clearly the largest, and the indoor PAEC$_{212}$ is comparable to the outdoor PAEC$_{222}$. Since - as noted in ref. 64 and discussed more thoroughly in Chapter 7 - a given PAEC from the ^{220}Rn series is thought to have much less significance than a comparable PAEC from the ^{222}Rn decay products, it appears that indoor concentrations of the 220 series are less significant even than outdoor radon concentrations. Nonetheless, the possible health significance is not to be ignored, and - indeed - understanding better the behavior of the ^{220}Rn series presents not only an interesting scientific problem, but even a tool by which to explore more fully the mechanisms for transport of the radon isotopes into the indoor environment.

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