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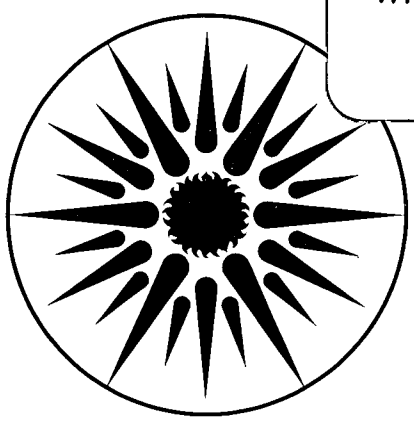

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ABSTRACT

Monitoring efforts undertaken before 1985 indicate that the radon concentration in U.S. houses averages approximately 55 Bq m^{-3} and that, in approximately 6%, annual-average levels exceed 150 Bq m^{-3}, with perhaps 1-2% having 300 Bq m^{-3} or more. However, several recent large-scale data sets yield average concentrations of 100-150 Bq m^{-3}, with perhaps 20% of results exceeding 150 Bq m^{-3}, leading many to conclude that the U.S. frequency distribution is substantially higher than previously indicated. In fact, this conclusion is unfounded, and the public, policy makers, and even scientists are being misled by inappropriate use of data. In particular, these recent data sets overrepresent high-concentration houses or include sampling performed in basements or in the winter only. Adequate information is not available to adjust these results precisely to annual-average indoor concentrations experienced by the population, but plausible corrections yield results that are consistent with the distributions previously found.

Substantial efforts have been devoted to monitoring of radon concentrations in U.S. homes in the last decade. The bulk of these efforts have occurred since the December, 1984 discovery of exceptionally high concentrations in the area of eastern Pennsylvania characterized geologically as the "Reading Prong". However, even before that discovery and the ensuing storm of concern and effort, significant monitoring efforts had already been undertaken, providing a substantial indication of the radon concentrations occurring in U.S. homes. The results of these earlier studies constitute a reasonable basis for considering the scope of the radon problem in the United States and, hence, for formulating an effective strategy for controlling excessive exposures to radon decay products.

However, the Pennsylvania discoveries changed the situation with regard to the perceived urgency of the problem, the level and nature of the effort devoted to it, and the manner in which results have been used. No better example can be found of this dichotomy between before and after the Reading Prong than the nature and use of results from indoor monitoring. Data sets developed since 1985 include thousands or tens of thousands of results, in contrast to the much smaller samples of houses monitored earlier, where individual studies monitored tens, or in rare cases hundreds, of homes. Ironically, these smaller sets still give us a better appreciation of the frequency distribution of radon concentrations in U.S. homes than the much larger data bases that are now available and receiving a great deal of attention, but that are not suitable for determining the concentration distribution, as discussed below.

EARLIER STUDIES

Results from the multiplicity of small studies were used in 1984 as the basis for developing a tentative U.S. concentration distribution.[1] This analysis handled the diverse natures of these studies by 1) discriminating between those undertaken because high indoor concentrations were known or suspected and those that were not, 2) normalizing data taken in the winter to an approximate annual-average concentration, since this is the result of principal interest for exposure (and risk) assessment, and 3) weighting the different data sets by number of houses, region, and population, to examine the sensitivity of the resulting distribution. The resulting aggregate distribution, found to be insensitive to weighting, has an average (AM) of 55 Bq m^{-3} (1.5 pCi/l), with an uncertainty - judged only internally from the data - of about 15%; because the various studies did

not employ a consistent methodology to select representative houses for monitoring, the real uncertainty is larger, but by an amount that cannot be estimated except on the basis of judgment. Approximately 2% of homes were found to exceed annual-average concentrations of 300 Bq m^{-3} (8 pCi/l). The distribution was represented well by a lognormal function with a geometric mean (GM) of 33 Bq m^{-3} (0.9 pCi/l) and a geometric standard deviation (GSD) of 2.8.

An interesting additional result was that - based on 4 sets of houses in which measurements were performed in both winter and summer - the annual-average concentration was, on the average, 72% of that measured in the winter. Further, the inclusion of data from areas where concentrations were expected to be higher on the basis of prior information raised the average concentration substantially, as expected. It is important to note that this analysis, as well as the distribution discussed next, utilized data accumulated in living space, not in basements, which is a major distinction from the recent, large data sets that have received the most attention.

A second important study entailed year-long radon monitoring in 1984-1985 in the homes of 453 physics faculty at 101 universities throughout the United States.[2] The results correspond well to a lognormal distribution with a GM of 38 Bq m^{-3} (1.03 pCi/l) and a GSD of 2.36; the AM was 54 Bq m^{-3} (1.5 pCi/l). This lognormal function, and that corresponding to the analysis discussed above, are displayed among the cumulative probability plots of Figure 1 as the two lower straight lines. In addition to similarity in AM, the Cohen result yields the fraction of houses having above 150 Bq m^{-3} (4 pCi/l) as 6%, as compared with 7% from Nero et al. This consistency in average and fraction above 150 Bq m^{-3} is reassuring, although it must be recognized that the virtual identities are fortuitous. Because of the difference in GSDs, perhaps due to reliance on a particular subpopulation (faculty) in Cohen's survey, the functions diverge at higher concentrations; still, the fractions above 300 Bq m^{-3} are only a factor of two apart (see Figure).

RECENT DATA BASES AND SURVEYS

An extremely large set of data that has received substantial attention is that from etched-track detectors. The results from approximately 60,000 U.S. measurements are found to average 266 Bq m^{-3} (7.2 pCi/l).[3] However, 50,000 of these results results arise from only 6 states, includ-

ing some where monitoring programs have been undertaken because of the existence of high concentrations. Removing these states yields an average, for the 10,251 remaining results, of 158 Bq m^{-3} (4.26 pCi/l). Two alternative methods of obtaining a more representative selection or weighting yield averages of 152 and 153 Bq m^{-3}. The authors find this convergence striking, assert that the results provide a rough "exposure" estimate, and conclude that the national "exposure" exceeds that cited previously, e.g., the 55 Bq m^{-3} average from Ref. 1. In point of fact, even the three restricted sets of data are likely to have an irremovable overrepresentation of homes with high concentrations. Further, the parameter of interest for indoor exposures is the annual-average concentration in the living space. We here indicate the probable scale of error in using the results of Ref. 3 as suggested there.

A major difficulty is that a large portion of the measurements are performed in basements: many U.S. homes have basements, and the protocols recommended, e.g., by the U.S. Environmental Protection Agency (EPA), for "screening" measurements suggest monitoring in basements where they exist. However, concentrations in basements average approximately twice those on first floors during the winter (with an even larger ratio during the summer).[4-5] In fact, Ref. 3 describes results from a subset of their measurements for which location was recorded, finding that 44% were taken in basements and have an average concentration 2.0 times the average from the non-basement readings. If we attempt to correct an apparent average of 155 Bq m^{-3} (4.2 pCi/l) to a living-space average assuming that 44% have results that are a factor of 2 high, we find a corrected average of 108 Bq m^{-3} (2.9 pCi/l), a large change. It is even more difficult, and probably impossible, to quantify the effect of oversampling in areas or houses where high concentrations are known or suspected. We note that a 25% reduction of the AM yields 81 Bq m^{-3} (2.2 pCi/l); as will be discussed elsewhere, there are indications from the etched-track data themselves and from comparison with the EPA data discussed below that the effect of oversampling is at least this large. Finally, due again to the nature of monitoring programs and recommendations, a preponderance of measurements are performed during winter only. As indicated above, annual-average concentrations have been found to average approximately 72% of winter values.[1] Given probable mixes of seasons in which the etched-track measurements were taken, the annual-average concentration may therefore be 80 or 90% of the value above, i.e., approximately 69 Bq m^{-3} (1.9 pCi/l). This is not in-

tended to represent an actual corrected value, because there is not enough information to adjust the results quantitatively. However, these considerations are enough to indicate that, based on the results from Ref. 3, an average less than 75 Bq m^{-3} (2 pCi/l) - and therefore consistent with the results discussed above - is probable, and that the values cited in the paper itself give a wholly misleading indication of public exposures.

Another major result of Alter and Oswald is that large numbers of homes have high concentrations, e.g., 23% of the 10,251 data in U.S.(-6 states) exceed 148 Bq m^{-3} (4 pCi/l), the EPA level above which remedial action is recommended. These results are plotted on Figure 1, where we see that a lognormal function with a GM of 61 Bq m^{-3} (1.65 pCi/l) and a GSD of 3.2 fit the points up to 148 Bq m^{-3} very well, but there is a very large excess of results with higher concentrations, especially at very high values. We only note that the considerations discussed above account for the fact that this curve is substantially above those of earlier studies, as well as for a substantially inflated percentage of houses exceeding 148 Bq m^{-3}.

Results of considerable interest have arisen from recent efforts of the EPA in conjunction with 10 states. During the winter of 1986-1987, charcoal radon detectors were deployed in statistically chosen samples of homes in nine states, plus another state in which volunteers were relied on. Examining the EPA press material[6], one finds the following information: the number of homes monitored in the main sample varied from 190 to 1787 for the ten states, totaling approximately 10,000. (Additional samples were taken on a volunteer basis, in at least one case to investigate areas thought to have a potential for higher concentrations than average.) The average concentration for the main sample was 110 Bq m^{-3} (3.0 pCi/l); 20% of measurements exceeded 148 Bq m^{-3} (4 pCi/l) and approximately 1.1% exceeded 740 Bq m^{-3} (20 pCi/l). Measurements were performed following the EPA screening protocol so that, as indicated in some of the EPA backup material, these results - taken in winter and, often, in basements - do not represent annual-average concentrations to which people are exposed. Nonetheless, the press reports, following the EPA press release, compared these results directly with the EPA action guideline and indicated that 21% (of the total) of 11,600 homes exceeded the guideline, an apparently inappropriate use of the data.

The data, however, promise to be very useful in other respects. On Figure 1 are plotted the probabilities corresponding to 148 and 740 Bq m^{-3}.

The lognormal function passing through these points has a GM of 58 Bq m^{-3} (1.56 pCi/l) and a GSD of 3.0, for which the corresponding AM is 106 Bq m^{-3} (2.9 pCi/l). This is close enough to the actual AM to suggest, based on the limited information in Ref. 6, that this lognormal function may be a useful representation of the data. If so, it is interesting to note that lognormality is maintained at least up to the region of 740 Bq m^{-3} (20 pCi/l). This contrasts strongly with the data of Ref. 3 and others who report raw or uncontrolled data, but consistent in this respect with the results from Refs. 1-2. That the distribution as a whole is higher than the earlier results is not surprising. As an illustration, if the same correction from basement concentrations as above is made to the AM of 110 Bq m^{-3}, and the resulting 76 Bq m^{-3} is corrected to an annual-average using the factor of 0.72 from Ref. 1, an AM of 55 Bq m^{-3} is found. The precise agreement with previous results is fortuitous, but indicates that - <u>if proper adjustments could be made</u> - agreement would be quite satisfactory. Furthermore, these adjustments would drastically reduce the fractions above 148 and 740 Bq m^{-3}, apparently to the vicinity of previous results, which were that 6 or 7% of homes exceed 148 Bq m^{-3} and perhaps 0.1% exceed 740 Bq m^{-3}.

The EPA data also give the first useful indication of the shape of the distribution at extreme concentrations, at least for these ten states. Extrapolating the lognormal representation to high levels indicates that approximately 0.01%, i.e., one home, ought to have exceeded 3700 Bq m^{-3} (100 pCi/l). The number actually observed is three, which is reasonably close. In terms of annual-average concentrations in living spaces, one expects from this data set that fewer than 3 in 10,000 homes exceed 3700 Bq m^{-3}. This contrasts strongly with the results of Ref. 3 and others who suggest much higher fractions exceeding this level.

This is not to say that the extreme tail of U.S. data does not exceed a lognormal representation. The ten EPA states may turn out to have less variance among them than the U.S. as a whole. Further, even one large area of high concentrations, such as that in the vicinity of the Reading Prong or - in England - that of Cornwall, can influence the extreme tail very substantially. But - statistically - a single such area might or might not have occurred, so that neither an excess nor a deficit in the extreme tail would be surprising, even if there are fundamental reasons to expect lognormality. On the other hand, smaller areas of high concentrations (such as those found in the Red River Valley and the Spokane River Valley)

probably have too few homes to influence the distribution as a whole in either direction. The outcome is that we do not yet have a reliable estimate of the extreme tail; in fact, we may never have, if only because, once found to have an extreme concentration, a house is typically modified before a long-term measurement in the living space is made. This leaves us with trying to estimate the extreme tail based on measurements that need adjustment; since - for a GSD of 3 and concentrations more than about 5 times the average - a factor of 2 difference in level changes the fraction of homes above that level by roughly an order of magnitude, such corrections, properly made, can drastically change estimates of the number of people exposed to such high levels.

OTHER STUDIES AND ADDED COMMENTS

Other monitoring efforts deserve discussion, but in this brief summary can hardly be mentioned. Cohen has developed a large data base of results from charcoal monitoring, including more than 30,000 measurements in living spaces;[4] the average concentration is approximately 136 Bq m^{-3} (3.6 pCi/l), but a subset of measurements offered gratis to randomly sampled individuals in a number of U.S. counties yields a substantially lower average. Of substantial interest from state or regional efforts are results, now becoming available, of a representative survey undertaken in approximately 2000 New York houses (an effort whose inception, again, predates the Pennsylvania discoveries), including year-long measurements in the living space; Figure 1 displays the four points that can be extracted from preliminary summaries of the data.[5] Major monitoring efforts have also been undertaken in other states, including more than 20,000 measurements by the Bonneville Power Administration (included in the etched-track data[3]) and thousands of measurements by the states of New Jersey and Florida.

An interesting aspect of available results taken as a whole is that they sometimes offer the opportunity of exploring dependencies on season and indoor location even when inadequate for indicating population exposures. In addition, correlations with influencing factors, such as soil characteristics, house type, and meteorology, can often be examined with such data, sometimes better than if a good exposure measure were obtained. Of more relevance to the subject of this paper, however, is that intercomparisons of different data sets sometimes improve our understanding of the nature and limitations of the individual sets themselves. Further

use of such intercomparisons will be described elsewhere. For the present purpose, it is enough to recognize that recent data, if properly interpreted, are consistent with previous results and, in significant respects, can provide new insights concerning the U.S. distribution of radon concentrations.

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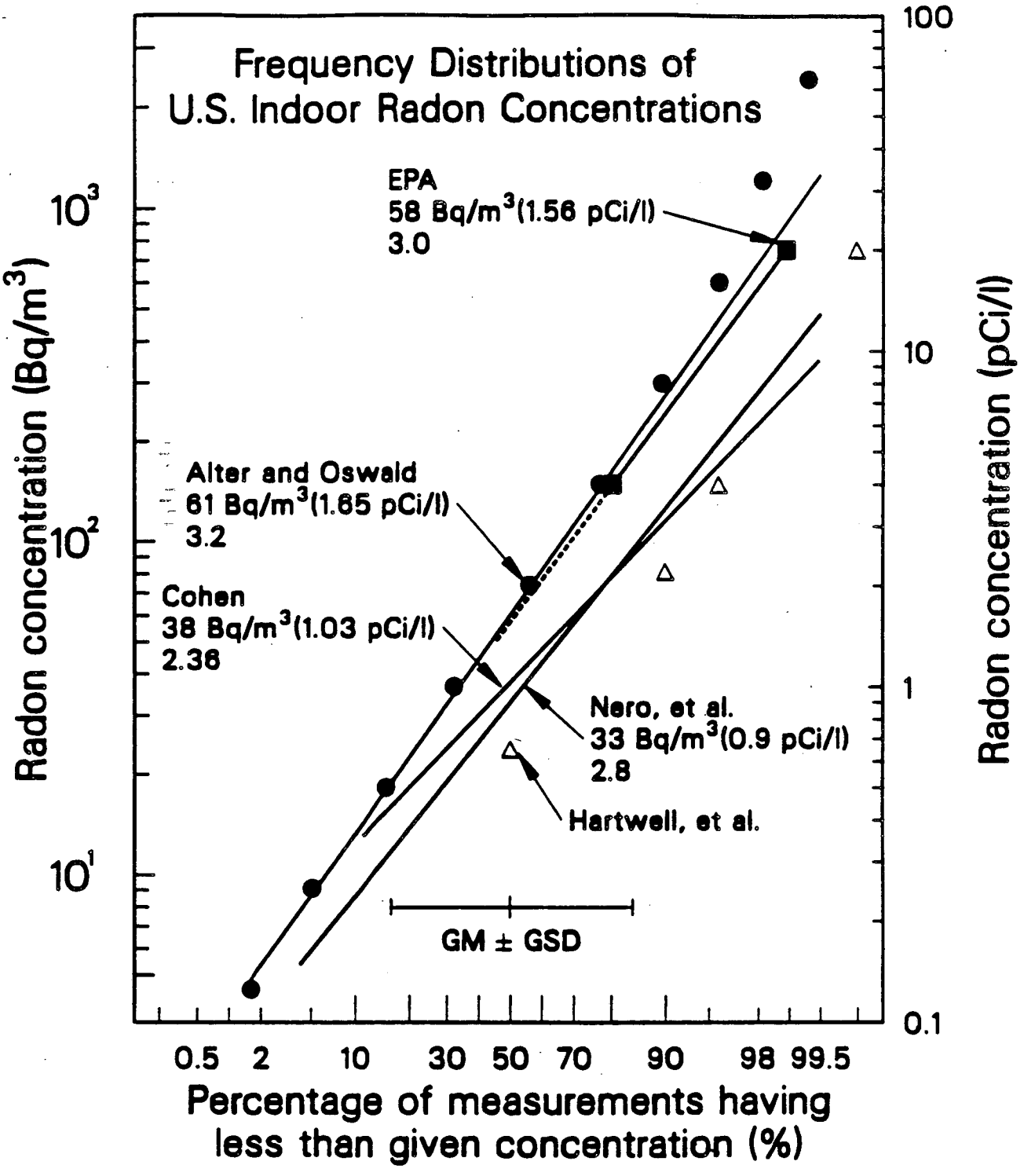

Figure 1. U.S. Radon Distributions. GMs and GSDs are given for each of the lognormal functions; see text for discussion.

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